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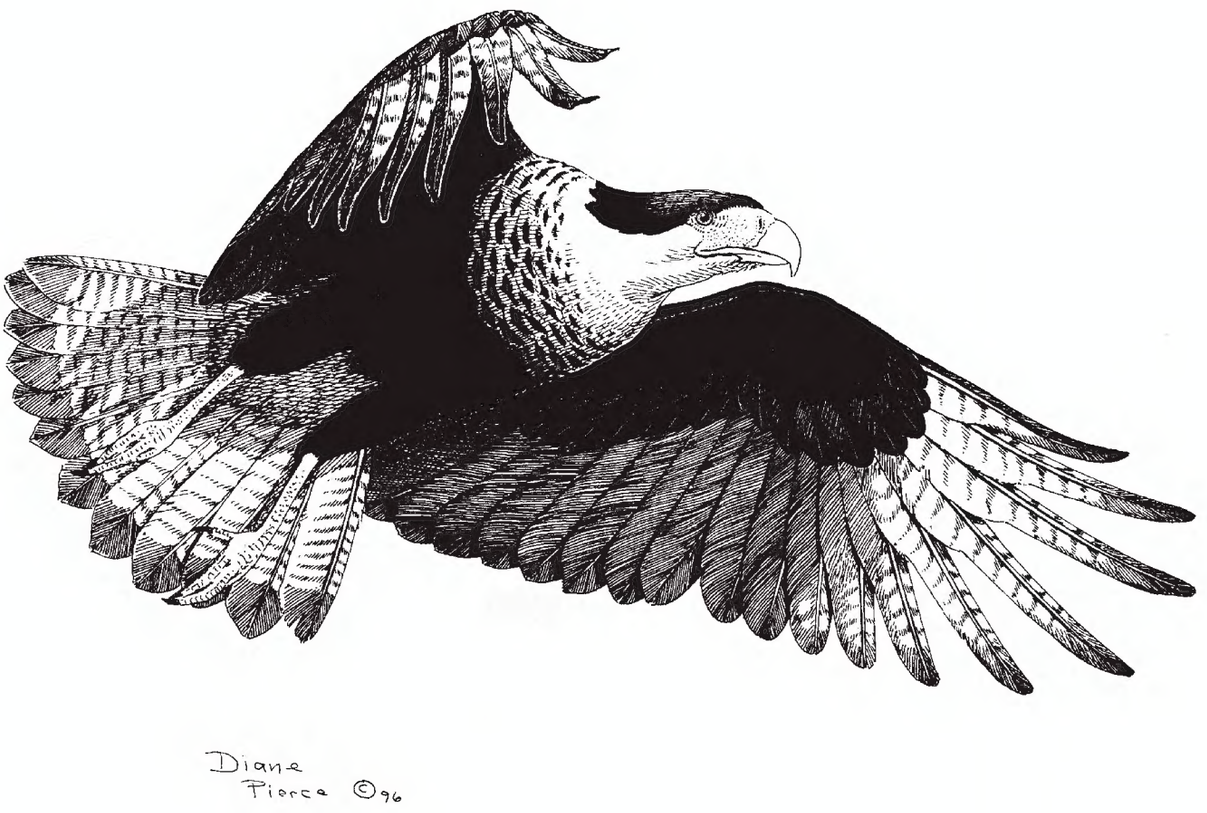

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SITE CHARACTERISTICS AND REPRODUCTIVE SUCCESS OF ROOF-NESTING AMERICAN OYSTERCATCHERS (*Haematopus palliatus*) IN FLORIDA

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Abstract.—American Oystercatchers (*Haematopus palliatus*) have adapted to urbanization of barrier islands in Florida by nesting on gravel-roofed buildings. This novel nesting habitat may offer an opportunity to conserve oystercatchers by providing refuge from human disturbance and sea-level rise, or it may provide low-quality habitat that serves as a population sink. This study aimed to describe site characteristics and determine hatching and brood success of roof-nesting American Oystercatchers in Pinellas County, Florida. We monitored roofs with a known history of nesting from 2017–2020, and documented nesting on 18 roofs. Nesting roofs varied in area and height but were located within 108 m of water, mostly the Gulf of Mexico or the Intracoastal Waterway. Hatching success (0.76) was higher than published ground-nesting estimates, whereas brood success (0.53) was comparable. Elevated hatching success in roof-nesting American Oystercatchers demonstrates a benefit to roof-nesting; however, this may be offset by high chick mortality after leaving the roof. Eight of 16 fledglings died or vanished upon leaving the roof, likely because of their lack of experience navigating the urban environment. Conservation of populations in developed areas may be possible through creation of nesting habitat on roofs immediately adjacent to beaches, but monitoring fledgling survival is imperative.

Key words: alternative habitat, behavioral plasticity, human disturbance, nesting success, rooftop, urban adaptation

Anthropogenic habitat loss is a major driver of declining biodiversity around the world, and many bird populations have been negatively affected (Dirzo et al. 2014). Habitat loss can be a result of direct conversion of land use (e.g., deforestation), or deterioration of habitat suitability by other anthropogenic pressures such as invasion

by exotic species, altered ecosystem processes, or disrupted disturbance regimes. For beach-nesting birds, human development, recreation, and disturbance renders nesting habitat inhospitable, a long-standing conservation concern (Burger 1995). Behavioral plasticity is an important mechanism enabling species to withstand such widespread anthropogenic changes (Wong and Candolin 2015).

In urbanized areas with high levels of human disturbance, gravel roofs offer alternative nesting habitat for beach-nesting birds. This behavioral adaptation has been documented in North America and Europe since the mid-1900s, primarily in Larids (Fisk 1978). Selection of roofs as nest sites in beach-nesting birds is believed to be a response to urban development, habitat degradation, and human disturbance in their traditional nesting habitat (Krogh and Schweitzer 1999, Douglass et al. 2001, Forys and Borboen-Abrams 2006). Roof nesters may gain relative solitude compared with beach nesters but also face unique challenges, such as chicks falling off unprotected roof edges, and flooding events during heavy rainfall on impervious flat roofs. In Florida, Least Terns (*Sternula antillarum*) have been nesting on roofs for over 60 years (Goodnight 1957), and American Oystercatchers (*Haematopus palliatus*) for at least 30 years (Paul 1988, Douglass et al. 2001). More recently, American Oystercatchers have been discovered nesting on roofs in urban areas of New Jersey (McConnell 2018) and South Carolina (F. Sanders, South Carolina Department of Natural Resources, pers. comm.), possibly indicating increased pressure on ground habitat along the Atlantic Coast of the United States. Eurasian Oystercatchers (*H. ostralegus*), which are more generalist in nesting habitat, also commonly nest on roofs in Europe (Bourne 1975, Munro 1984, Duncan et al. 2001).

Across their range, American Oystercatchers have demonstrated flexibility in habitat selection; sites such as dredge spoil islands and shell rakes are now vital nesting habitat (Toland 1992, Virzi 2008, Jodice et al. 2014). Reproductive success in these alternative habitats has been found to be higher than at traditional barrier island sites (McGowan et al. 2005, Virzi et al. 2016), where human activity and associated mammalian populations are abundant. Given the continued human population growth along the Atlantic Coast of the United States, alternative nesting sites are likely to become more important to American Oystercatchers, but these habitats are also declining because of sea-level rise, loss of oyster reefs, and erosion (Vitale et al. 2020).

American Oystercatchers are a species of conservation concern throughout much of their range, including Florida where they are listed as Threatened (Florida Fish and Wildlife Conservation Commission [FWC] 2013). Human disturbance and development of nesting habitat resulting in low productivity is a major threat (McGowan and Simons

2006, Sabine et al. 2008, Virzi 2010, Borneman et al. 2016), as is erosion of nesting habitat (Vitale et al. 2020), and much of the management of this species has focused on increasing productivity (American Oystercatcher Working Group et al. 2012, Felton et al. 2017). Although the plight of ground-nesting American Oystercatchers has been well documented, little is known about roof nesting in this species, most notably habitat selection and reproductive success.

The purpose of this study was to better understand roof-nesting in American Oystercatchers and to evaluate roofs as an opportunity to conserve American Oystercatchers within urban environments. Specific objectives were to describe characteristics of roof nest sites, and to determine hatching and brood success of roof-nesting American Oystercatchers in Florida.

METHODS

Pinellas County is the only place in Florida where American Oystercatchers have been documented nesting on roofs in recent years. Pinellas County, the most densely populated county in Florida, is located in west-central Florida between Tampa Bay and the Gulf of Mexico (Fig. 1). Nearly uninterrupted development spans 39 km along its barrier islands.

The author and trained volunteers monitored 92 gravel roof sites in Pinellas County with a known history of roof-nesting by beach-nesting birds (American Oystercatchers, Least Terns, or Black Skimmers [*Rynchops niger*]) from 2017–2020 for nesting activity by American Oystercatchers. At roofs with nesting oystercatchers, I documented the following site characteristics: roof height (stories), roof area, presence of a parapet (wall around the perimeter of roof), building type (residential or commercial), distance to nearest water, and shoreline type (seawall, beach, mangrove, or riprap). I also obtained data on 9 other roofs with a previous history of American Oystercatcher nesting in Florida from 2011–2016 to describe site characteristics (E. Forys, Eckerd College, unpublished data). I used Google Earth Pro 7.3.2.5776 software to measure roof area and distance to water at each roof-nesting site.

The author and volunteers conducted 15-minute observations at each site approximately every two weeks from March through July to identify the presence of American Oystercatchers following FWC protocol (FWC 2020). Once we identified American Oystercatcher activity, we visited roofs at least weekly to monitor nesting through completion (n = 943). To avoid disturbing birds, we monitored nesting from the ground or an adjacent structure that offered a vantage onto the roof when available. For analysis of hatching and brood success, I included only sites with a suitable vantage where we could observe incubation and chick-rearing activities with the aid of 10×42 binoculars or 22–48×65 spotting scopes.

I calculated hatching and brood success using the Mayfield method (Mayfield 1975), with an assumed 27-day incubation period and 35-day fledging period. I calculated 95% confidence intervals (CI) following Johnson (1979). I defined hatching success as ≥1 chick hatching from a clutch, and brood success as ≥1 individual of a brood reaching 35 days old (hereafter fledge-age). When nesting attempts failed near the time of hatching and we never observed chicks, I assumed failure occurred during the egg stage. American Oystercatchers are single-brooded renesters, and renesting attempts were included in the analysis. We closely monitored the fate of fledge-age chicks (approximately four times per week) to the extent possible by searching adjacent roofs, beaches, and foraging areas

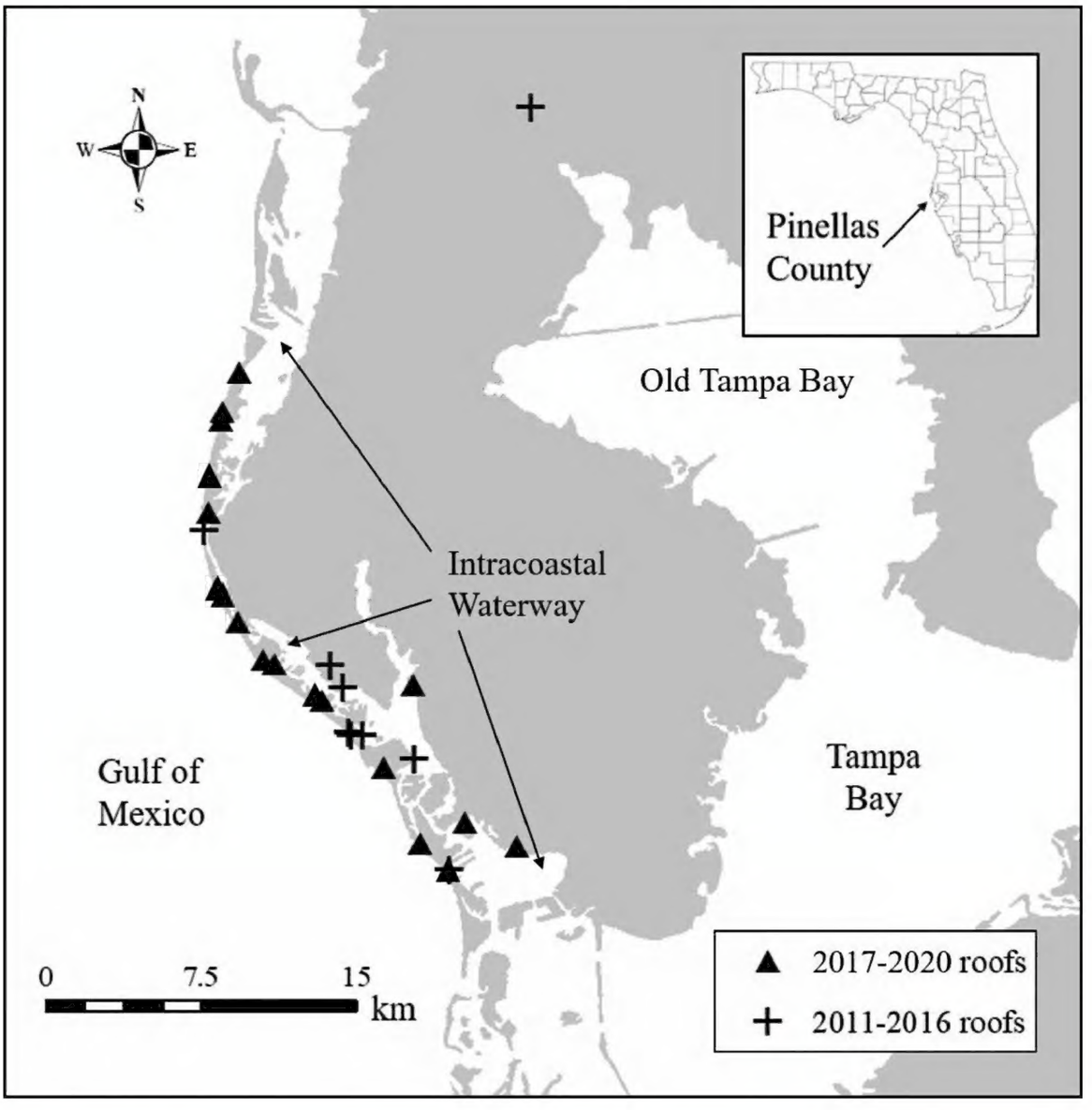

Figure 1. American Oystercatcher (*Haematopus palliatus*) roof nest sites in Pinellas County, Florida, USA.

in the seven days following departure from the roof. I used the FWC definition of productivity rate: the number of fledglings divided by the number of breeding pairs each year (FWC 2013).

Results

From 2017–2020, we identified American Oystercatcher nesting at 18 roof sites, including five opportunistically discovered roofs with no known history of nesting. We observed nesting activity between 9 March and 1 August, with a single pair of American Oystercatchers occupying each roof, although sometimes in association with Least Terns, Black Skimmers, and Killdeer (*Charadrius vociferous*).

Roofs with American Oystercatcher nesting from 2011–2020 (n = 27) had a mean area of 1,434 m^2 (± 318 SE), but nesting occurred on

roofs as small as 77 m^2 (range = 77–7,284 m^2). Buildings ranged from 1–8 stories in height (median = 2), and the majority of the buildings were residential (18 apartments or condominiums, 3 single-family homes). Commercial buildings were retail (3), industrial (1), hotel (1), and school (1). Ten roofs (37%) were completely encompassed by parapets. Roofs ranged 2–108 m from water (mean = 36.7 ± 7.1 SE), and all but one were located along the Gulf of Mexico or Intracoastal Waterway (Fig. 1). Shoreline types for the nearest waterbody were seawall (15), beach (9), mangrove (2), and riprap (1).

Twelve different roofs had a suitable vantage for monitoring nest fate in 2017–2020, with 30 nesting attempts monitored. Across all years, daily nest survival and daily brood survival were 0.990 (n = 30) and 0.982 (n = 23), respectively. Overall hatching success was 0.76 (95% CI = 0.62–0.93), and brood success was 0.53 (95% CI = 0.45–0.82). Chicks remained on roofs until approximately 35–45 days of age. The 30 nesting attempts resulted in 16 fledge-age chicks on roofs, and a productivity rate of 0.76 chicks/pair (Table 1). Eight of the 16 fledge-age chicks were confirmed to have survived for at least one week after leaving the roof, one died after a collision with a sign, and three others died as determined by immediate re-nesting of the pair. In the remaining four cases, we never observed chicks off the roof, and believed they died shortly after leaving the roof because we observed a pair of adult American Oystercatchers in the vicinity of the roof without a chick. Additionally, roofs without a suitable vantage to monitor hatching success fledged six chicks during the four-year study period.

DISCUSSION

This study provides the first quantitative information on the productivity of roof-nesting American Oystercatchers, and demonstrates the potential for gravel roofs to be used as a tool for conserving American Oystercatchers in developed landscapes. Characteristics of roofs selected by American Oystercatcher pairs varied considerably in building height and roof area, from one-story single-family homes to high-rise condominiums and supermarkets. Site tenacity at roofs is high, with some sites occupied continuously for 13 years (E. Forys, unpublished data). All but one of the American Oystercatcher roofs were located close to the Gulf of Mexico or the Intracoastal Waterway, indicating close foraging areas as an important element of site selection. The exception was a roof adjacent to a canal that leads into Old Tampa Bay. Roof-nesting Least Terns also select buildings of various heights and roof areas but, unlike American Oystercatchers, sometimes select buildings at great distance from large bodies of water (Forys and Borboen-Abrams 2006).

Table 1. Annual summary of American Oystercatcher (*Haematopus palliatus*) roof-nesting in Pinellas County, Florida, 2017–2020. Visible roofs had a suitable vantage for monitoring reproductive success without accessing the roof. I considered chicks to be fledge-age at 35 days. Productivity rate is fledglings per pair.

Year	Roofs with nesting	Visible roofs	Fledge-age chicks on visible roofs	Fledglings surviving ≥1 week after leaving roof (visible roofs)	Productivity rate (visible roofs)	Fledglings surviving ≥1 week after leaving roof (all roofs)	Productivity rate (all roofs)
2017	8	2	1	1	0.50	3	0.38
2018	8	7	6	3	0.42	3	0.38
2019	7	7	3	0	0.00	0	0.00
2020	11	5	6	4	0.80	8	0.73
Total	34	21	16	8	0.38	14	0.41

Although I did not directly access roofs to avoid disturbance, and did not formally assess microhabitat characteristics, nests appeared to be preferentially placed within 0.25 m of a roof vent or other small feature on the roof but far from larger objects like air conditioner condensers or tall parapets. This placement could help make exposed eggs less conspicuous to aerial predators while still allowing clear sightlines for incubating adults (Buckley and Buckley 1980, Lauro and Nol 1995).

Young chicks spent much of the time out of sight underneath air conditioner condensers or similar structures, venturing out only for a few moments to receive food provisions from a parent. In 2019 and 2020, I placed wooden A-frame and pallet-style chick shelters on six roofs to provide chicks protection from sun and predators. Chicks of all ages readily used shelters when I supplied them at sites with little cover available.

Hatching success of roof-nesting American Oystercatchers was higher during this study than reported hatching success of ground-nesters (usually 0.11–0.45; Davis et al. 2001, McGowan et al. 2005, Sabine et al. 2006, Koczur et al. 2014, Schulte and Simons 2015, Virzi et al. 2016). The most plausible explanation for increased hatching success on roofs is the lack of overwash and mammalian predation, the leading causes of nesting failure in ground-nesting American Oystercatchers (American Oystercatcher Working Group et al. 2012). Additionally, actual roof-nesting hatching success may be higher than the estimate in this study because of the assumption that failure occurred during the egg stage when we never observed chicks. My results are similar to those found for roof-nesting Eurasian Oystercatchers (Duncan et al. 2001), and Least Terns (Gore and Kinnison 1991, Krogh and Schweitzer 1999), indicating a benefit to roof nesting across species.

Brood success of American Oystercatchers nesting on roofs was comparable to most published estimates for ground-nesting American Oystercatchers (0.33–0.65; Thibault et al. 2010, American Oystercatcher Working Group et al. 2012, Schulte and Simons 2015, Virzi et al. 2016). The only confirmed cause of mortality during chick rearing was a case of human disturbance, but Fish Crow (*Corvus ossifragus*) predation was strongly suspected in other cases where adults were repeatedly fending off crows in the days prior to failure.

The high mortality rate experienced by fledge-age chicks leaving the roof is attributable to hazards posed by the urban environment. Chicks learning to fly struggled to safely navigate obstacles and we observed them crashing into posts and bushes shortly after leaving the roof. Additionally, residents of buildings with roof nesting reported chicks dying in previous years from drowning along seawalls and being hit by automobiles. The productivity rate of 0.76 fledge-age chicks per pair on visible rooftops exceeds the FWC's goal of 0.50 fledglings/pair

for population growth (FWC 2013). The realized productivity rate accounting for the observed mortalities (productivity rate × survival rate of fledglings upon leaving the roof), however, was only 0.38 fledglings/pair among visible roofs.

Gravel roofs offer urban nesting habitat that opportunistic American Oystercatchers are taking advantage of; however, development of superior roofing materials has made gravel roofs obsolete. As gravel roofs are replaced, this nesting habitat will be lost (DeVries and Forys 2004), displacing nesting American Oystercatchers. Availability of American Oystercatcher nesting habitat is a major concern in Florida (Vitale et al. 2020), and roof nesting offers a potential refuge from the threats of human disturbance and rising sea levels. As American Oystercatchers have demonstrated nesting on roofs as small as 77 m^2, it may be possible to retain nesting American Oystercatchers in urbanized settings by providing gravel nest boxes on top of buildings. Placing nest boxes on buildings immediately adjacent to beaches may offer a safer environment for fledging chicks compared to roofs located along seawalls, but monitoring the fate of fledglings will be necessary to determine the conservation benefit of roof nesting.

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RECORD OF A RED-RUMPED AGOUTI (*Dasyprocta leporina*) IN MIAMI-DADE COUNTY, FLORIDA

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To be authoritative and to document the history of a species, faunal lists should include taxa that escape or are released from captivity. Florida is a hotspot for exotics because of its semi-tropical climate, the diversity of its natural and human-modified landscapes, its large, ever-increasing, and diverse human population, and its proximity to other regions (Simberloff et al. 1997). Mammals represent one of the least diverse groups of exotics in the state, and available information is scarce: Layne (1997) focused on 22 species with then-currently or formerly established populations, along with four domesticated species with free-roaming populations. The Florida Fish and Wildlife Conservation Commission's (FWC 2020) website on exotic mammals likewise excludes many species. Given the scarcity of knowledge on Florida's exotic mammal fauna, any documented record would be significant.

During the weekend of 13–14 June 2020, we were birding in southeastern Florida. At 0855 hours on 14 June 2020, as we were traveling west on SW 256th Street east of its intersection with SW 187th Avenue in the Redland area west of Homestead, Miami-Dade County, we saw an unfamiliar mammal amble across the road. It was the size of a small dog, had virtually no tail, and had dull brown pelage with a brighter, rufous hind end. BP took several photographs of the animal as it trotted away, pausing briefly twice before it moved into dense vegetation and out of our view. After ruling out canine species, we settled on some species of large rodent, eventually settling on red-rumped agouti (also known as Brazilian agouti; *Dasyprocta leporina*). The best and clearest photograph (Fig. 1) allows the animal to be sexed as a male. Red-rumped agouti is widespread in a variety of forested areas in Trinidad and Tobago, the Lesser Antilles, and South America from Venezuela to northeastern Brazil. The land use of the location of our sighting was tropical fruit groves such as mango (*Mangifera* spp.), interspersed with a few residences and agricultural buildings.

The only information we have found on red-rumped agoutis in Florida is from the EDDMapS (Center for Invasive Species and Ecosystem Health 2020) and iNaturalist (2020) websites, which collectively map seven additional records, all of them single individuals, and all but one occurring within the past 10 years. From north to south, the records are from Wesley Chapel, Pasco County, on 8 September 2020; Lutz, Hillsborough County, on 25 May 2020; Tampa, Hillsborough County, on 16 August 2020; an unstated location in Hillsborough County on 28 March 2020; southeast of Plant City, Hillsborough County, on 1 September 2007; and separately in Miami, Miami-Dade County, on 15 May 2012 and 3 September 2015. Our sighting provides at least the eighth record for Florida and the third record from Miami-Dade County.

We thank Elsa Alvear for her gracious hospitality, and Jane Widness for confirming the identification of the agouti on iNaturalist. Don Fraser and an anonymous reviewer improved drafts of the manuscript.

Figure 1. Red-rumped agouti at Homestead, Miami-Dade County, Florida, on 14 June 2020. Our sighting represents at least the eighth documented record in the state. Photograph by Bill Pranty.

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TWO NEW RECENT COUNTRY RECORDS OF MORMOOPID BATS (CHIROPTERA: MORMOOPIDAE) FROM LONG ISLAND, THE BAHAMAS

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The Caribbean is home to a large diversity of bats, spanning nine families and at least 60 genera. Within The Bahamas, there are 10 species of bats that have established populations: *Artibeus jamaicensis*, *Chilonatalus tumidifrons*, *Eptesicus fuscus*, *Erophylla sezekorni*, *Lasiurus minor*, *Macrotus waterhousii*, *Monophyllus redmani*, *Nyctiellus lepidus*, *Noctilio leporinus*, and *Tadarida brasiliensis*. There are also three that are documented as single specimens and were presumably vagrants: *Glossophaga soricina*, *Lasionycteris noctivagans*, and *Lonchorhina aurita* (Buden 1986, Speer et al. 2015).

The family Mormoopidae is not currently extant in The Bahamas but is represented in the fossil record. *Pteronotus macleayii* is known from New Providence (Morgan 1989), *P. parnellii parnellii* is known from New Providence and Great Abaco (Olson and Pregill 1982, Morgan 2001, Soto-Centeno and Steadman 2015), and *P. quadridens* is known from fossils on Great Abaco, North Andros, and New Providence (Morgan 2001, Dávalos and Turvey 2012). *Mormoops blainvillei* is known from fossils on Little Exuma, Great Abaco, and New Providence (Koopman 1951, Olson and Pregill 1982, Morgan 2001) and *M. megalophylla* was found on Andros and Great Abaco (Morgan 2001). Extant populations of all these species are found in the Greater Antilles, except for *M. megalophylla*, which is extirpated from the Caribbean (Dávalos and Turvey 2012).

Long Island, an inhabited island in the central part of The Bahamas archipelago, is approximately 130 km long and 6 km wide (Fig. 1). It is about 265 km southeast of the Bahamian capital of Nassau and is part of the Great Bahama Bank, which during the last glacial period (80–10 ka) would have formed one large island along with Andros, Bimini, the Berry Islands, Cat Island, Eleuthera, the Exumas, New Providence, and Ragged Island. It is the south-easternmost island of any significant size on the Great Bahama Bank. There are six species of bats that have known established populations on Long Island: *E. fuscus*, *E. sezekorni*, *L. minor*, *M. waterhousii*, *N. lepidus*, and *T. brasiliensis* (Speer et al. 2015).

On 6 January 2017, we were hand-netting bats in Great Lakes cave (22.92 °N, 74.86 °W), which is found on the south end of the island, approximately 22.5 km southeast of Clarence Town, near Mortimer's settlement. This cave has also been referred to as Mortimer's Cave by previous researchers (Speer et al. 2015). We were opportunistically sampling bats in this cave, and other caves throughout Long Island. As we were hand-netting near the entrance of the cave, we captured one individual each of the sooty mustached bat, *Pteronotus quadridens*, and the Antillean ghost-faced bat, *Mormoops blainvillei*, almost simultaneously within the first 10 minutes of netting (Fig. 2). The *P. quadridens* was an adult male with a forearm length of 40 mm and weighing 7 g. *Pteronotus quadridens* might most likely be confused with *P. macleayii*, but the short forearm length and small size indicated it was indeed *P. quadridens* (Pavan 2019). The *M. blainvillei* was an adult female with a forearm length of 47 mm and weighing 9.5 g. We easily identified this species because it is the only *Mormoops* in the Greater Antilles and much smaller than *M. megalophylla*, the only other species in the genus (Pavan

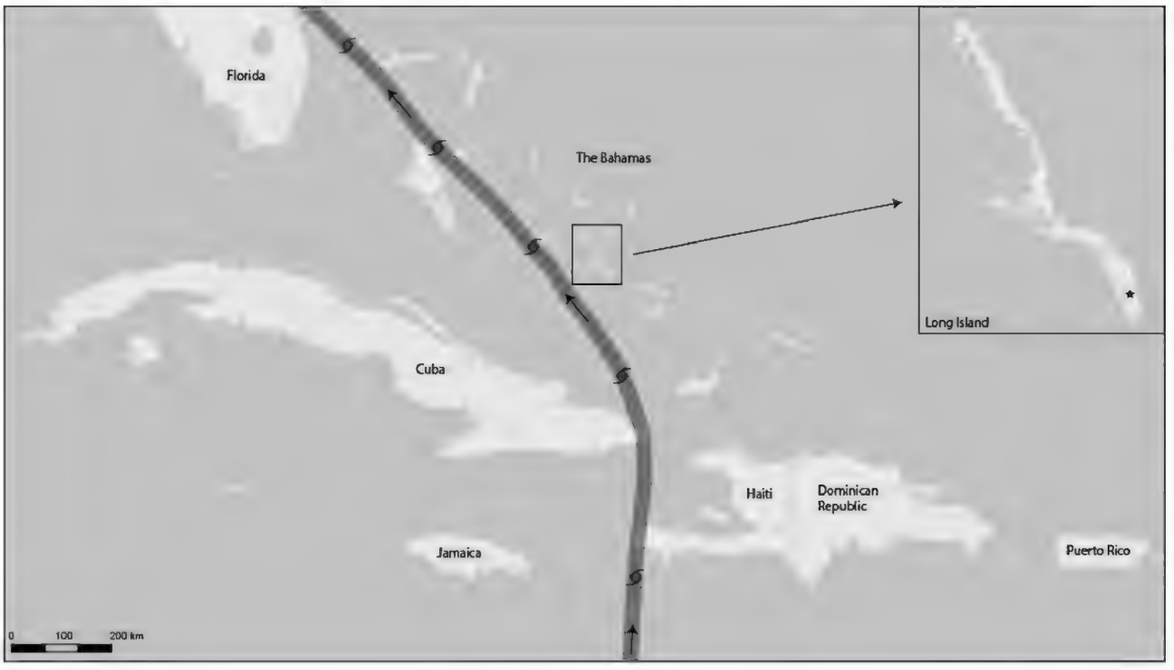

Figure 1. The Greater Antilles and The Bahamas. Black box is Long Island and inset in upper right is a close-up of Long Island with the location of Great Lakes cave indicated by the star. Gray line shows the general path of Hurricane Matthew in 2016 with directional arrows.

2019). Following capture, we sent photos of these bats to a colleague who confirmed our identifications (J. A. Soto-Centeno, pers. comm.). We also captured and released 11 *T. brasiliensis*, 17 *M. waterhousii*, and 7 *E. sezekorni* during the visit. We captured *Nyctiellus lepidus* during a visit to this cave in 2012, but we did not see it during this trip. We did not net or see any other species at that cave, and did not see any more individuals of the two newly recorded species in the several hours we were in the cave.

We hand-netted all bats sampled, and we recorded identification, sex, weight, forearm length, ear length, hindfoot length, and tragus length for each individual. We used a 4-mm biopsy punch to take a genetic sample from the wing membrane. We released all bats except the two new records after handling; we collected the *M. blainvillei* and *P. quadridens* individuals and deposited them in the Florida Museum of Natural History (catalog numbers UF 33956 and UF 33957, respectively). We handled all bats according to the guidelines for the use of wild mammals in research (Sikes et al. 2016). This research was approved by the University of Florida's Institutional Animal Care and Use Committee (protocol #201609641).

The southern end of Long Island is approximately 211 km from Cuba, the closest island in the Greater Antilles that has both *M. blainvillei* and *P. quadridens*. We believe the bats may have been brought from Cuba by Hurricane Matthew, which passed through the Caribbean in late September-early October 2016 (Fig. 1). Matthew was a Category 4 hurricane when it landed on Cuba and Category 3 when it passed through the southern Bahamas, strengthening to a Category 4 again by the time it passed the northern Bahamas with winds topping out at 145 mph. Both *M. blainvillei* and *P. quadridens* are typically known to roost in hot caves, although their presence as fossils in The Bahamas suggests that they may have been able to adjust to the cooler cave roosts found there (Soto-Centeno et al. 2015). Yet their current extirpation implies that the long-term conditions may not have been favorable for their continued presence. This, along with the fact that the distance between the islands is so great, and that two unusual species were found in the same cave at the same time, suggests that it is highly unlikely this was an intentional dispersal event.

Figure 2. The two individuals captured in Great Lake cave on Long Island, The Bahamas, in January 2017. Top: sooty mustached bat (*Pteronotus quadridens*). Bottom: Antillean ghost-faced bat (*Mormoops blainvillei*). Photo by VLM.

Hurricane-influenced disturbances on bat populations are not uncommon, where they can suppress populations, cause mortalities, and alter genetic diversity (Gannon and Willig 1994, Jones et al. 2001, Rodríguez-Durán and Vázquez 2001, Fleming and Murray 2009, Pederson et al. 2009). Further, there are several examples in the West Indies where a hurricane is the only likely explanation for the documented presence of unlikely bat species in new areas (Kwiecinski and Coles 2007, Gannon and Willig 2009, Pederson et al. 2009, Rivas-Camo 2020). Hurricane-assisted records such as these are important to document the role such storms may play in assisting species to colonize new areas and altering genetic diversity in existing populations.

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PLAY BEHAVIOR BY AN ANHINGA (*Anhinga anhinga*)

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The various behaviors called play, so prominent and widespread among humans, have also been observed in a growing number and variety of other animals (Burghardt 2005). Here I report a previously undescribed behavior of an Anhinga (*Anhinga anhinga*) that appears to qualify as play.

Sandra Marraffino of Rainbow Springs was kayaking near the headsprings of the Rainbow River, Marion County, Florida at about 1000 on 5 February 2017 when she observed an Anhinga repeatedly tossing a feather into the air. The Anhinga threw the feather upward and caught it three times (Figs. 1, 2). On the fourth toss the feather fell into the water and the Anhinga picked it up (Fig. 3). Marraffino then lost sight of the Anhinga as the current moved her kayak away (S. Marraffino, pers. comm.).

The Anhinga appears to have been in a plumage preceding the definitive adult feathering acquired in the third calendar year (Frederick and Siegel-Causey 2020). The feather it threw resembles a tail feather of an Anhinga.

This behavior exhibits some of the dozen or so characteristics used by various authors to define the term play. According to the most commonly cited of these, play seems to serve no immediate purpose, it is repeated, and it gives the impression of being voluntarily initiated (Burghardt 2005).

The types of play are commonly divided (e.g., Fagen 1981, Burghardt 2005, O'Hara and Auersperg 2017) into three broad categories: locomotor play, such as running and jumping; social play, such as chasing or mock fighting with others, usually conspecifics; and object play, such as picking up and manipulating inanimate materials. These categories are not mutually exclusive.

Object play has been "proposed to constitute an important factor in developing skills concerned with physical problem solving" (O'Hara and Auersperg 2017:119), which is to say, practice. Object play may be useful as well in developing skill at problem solving, especially in generalist large-brained and occasionally tool-using birds such as parrots and corvids (O'Hara and Auersperg 2017).

Play of one kind or another has been claimed for a wide range of non-human mammals (Burghardt 2005) including dolphins (DelFour et al. 2017) and domestic dogs (Bekoff 2015), many species of birds (Ficken 1977, Ortega and Bekoff 1987), fish (Burghardt 2015), possibly invertebrates such octopuses (Burghardt 2005), and even tyrannosaurs (Rothschild 2015). Play is generally more common among juveniles than adults (e.g., Biondi et al. 2013), and adult play seems to occur mainly in species with opportunistic foraging habits (O'Hara and Auersperg 2017).

Ortega and Bekoff (1987) found plausible descriptions of play among 13 orders of birds, including groups as varied as penguins, falcons, swifts, and songbirds. Among these, play was far more common among altricial rather than precocial species, and object play was the most common type reported.

Anhingas have not previously been reported to toy with feathers, but young Anhingas have been seen throwing and catching sticks and other plant material (Stevenson and Anderson 1994, Davis 2015). Object play has been reported a few times among relatives

Figure 1. An Anhinga on the Rainbow River, Florida, 5 February 2017, as it throws a feather in the air. Photograph by and courtesy of Sandra Marraffino.

of the Anhinga. Sazima (2019) saw an Australasian Darter (*A. novaehollandiae*), apparently a sub-adult, repeatedly throw and catch a "pod" (seed pod?). Young Great Frigatebirds (*Fregata minor*) pick up floating objects such as leaves and then drop them. Cormorants (*Phalacrocorax* spp.) likewise have been observed playing with fish, rocks (Ficken 1977), and various plant parts such as twigs and leaves (Sazima 2008; S. Marraffino, pers. comm), though not with feathers.

Several more distantly related species have been reported to play in ways similar to the behavior of this Anhinga. Ravens (*Corvus* spp.) drop bits of plants and catch them in the air (Ficken 1977). Swallows are known to play with feathers (Ficken 1977). Kasambe (2011) reported a River Tern (*Sterna aurantia*) repeatedly picking a stick from the water, dropping it, and retrieving it from the water.

Practice play incorporates movement patterns from real functional contexts (O'Hara and Auersperg 2017). The object play Marraffino observed is reminiscent of the way

Figure 2. The Anhinga catches the feather on the Rainbow River, Florida, 5 February 2017. Photograph by and courtesy of Sandra Marraffino.

Figure 3. The Anhinga retrieves the fallen feather from the water on the Rainbow River, Florida, 5 February 2017. Photograph by and courtesy of Sandra Marraffino.

in which Anhingas will toss a fish to catch it with the open bill before swallowing it (Frederick and Siegel-Causey 2020). This relatively young Anhinga may have been practicing the finely tuned movements needed to intercept an unpredictably falling object such as a fish.

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ABSTRACTS OF STUDENT PRESENTATIONS AND POSTERS AT THE FALL 2019 MEETING OF THE FLORIDA ORNITHOLOGICAL SOCIETY

2 November 2019
Institute for Water and Environmental Resilience
Stetson University
DeLand, Florida

The fall 2018 meeting of the Florida Ornithological Society featured a special set of oral presentations and posters prepared primarily by students associated with Florida universities. Many of the presentations will likely lead to full-length articles, but Abstracts submitted by participants are provided here for members of the Florida Ornithological Society who could not attend the meeting and to serve as a record of the breadth of research underway by the next generation of Florida scientists.

The oral presentations and posters sessions were organized by Peter Monte (Department of Wildlife Ecology and Conservation, University of Florida). Thanks to Peter for bringing the presentations together and to Shelley Gentile and Stetson University for their help organizing the meeting. The Abstracts are listed alphabetically based on the last name of the presenter (who is also listed as the email contact). All authors' affiliated institutions are in Florida unless otherwise noted.

Habitat selection and quality in Southeastern American Kestrels in Florida scrub

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Habitat selection theory often focuses on relationships between preference and its fitness outcomes, where theory has made a wide

array of predictions. Theory often assumes that individuals should behave adaptively by selecting the highest quality habitat. Conversely, human-induced environmental change that attempts to mimic natural processes may sometimes lead to maladaptive habitat selection behaviors where selection is decoupled from fitness. These behaviors may lead to an ecological trap, where individuals prefer poor-quality habitats, or undervalued resources, where individuals avoid high-quality habitats. The Southeastern American Kestrel (*Falco sparverius paulus*) is a species of conservation concern that uses intensively managed scrub maintained by prescribed burns and clearcutting, but the relationship of habitat selection and breeding fitness outcomes remains poorly understood. Our objectives were to 1) determine factors explaining kestrel occupancy (a proxy for habitat selection) and 2) assess the relationship between kestrel occupancy and fitness. We conducted point counts and nest searching and monitoring during the 2018–2019 breeding seasons in scrub habitat in Ocala National Forest, Florida. We used single-species occupancy models to determine habitat variables influencing kestrel occupancy. We used generalized linear models to determine factors influencing daily nest survival and productivity. Stand age (i.e., time since clearcutting or burning) and snag availability were the best predictors of kestrel occupancy. Probability of occupancy decreased as stands got older. Apparent nest success was high (69% of nests produced at least one fledgling), but daily survival was not related to stand age, even though kestrels are more likely to occur in younger stands. Consequently, kestrel occupancy did not correlate with daily nest survival or productivity. We discuss this pattern in the context of predictions from habitat selection theory, undervalued resources, and the challenges of interpreting whether fitness and habitat selection are decoupled.

The Grasshopper Sparrow genome

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Reference genomes are powerful tools in the field of conservation genomics because they enable researchers to link patterns and processes to specific traits within the genome. They can be applied to a host of questions not limited to evolutionary biology, disease, and population dynamics. For my master's research, I am assembling a first-generation

annotated reference genome of the eastern Grasshopper Sparrow (*Ammodramus savannarum pratensis*) and investigating the timing and extent of reproductive isolation between the eastern Grasshopper Sparrow and the Florida Grasshopper Sparrow (*A. s. floridanus*). Specifically, I am interested in how their differing migratory behaviors are involved in their divergent evolution and whether we can detect the timing and level of gene flow between populations. We sequenced a near-complete genome of *A. s. pratensis* and the transcriptome, which represents the sequences of expressed genes within the genome. Presently, I am producing a structural and functional annotation to identify genes and their functions via bioinformatics. In addition to the annotated reference, I am currently generating reduced-representation sequencing libraries of both sparrow populations that will be aligned to the reference. The combination of these data will provide insights into the specific traits and evolutionary processes involved in the divergence of the eastern and Florida Grasshopper Sparrow.

Red-cockaded Woodpecker cavities as potential ecological sinks for nesting avian kleptoparasites

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The endangered Red-cockaded Woodpecker (*Leuconotopicus borealis*) is a unique keystone excavator of live pines in mature southeastern forests. Cavities excavated by Red-cockaded Woodpeckers provide roost and nest sites for a suite of taxa in an otherwise cavity-poor environment, but cavity kleptoparasitism can threaten recovering Red-cockaded Woodpecker populations because of extended excavation time in live pines. We investigated conditions driving cavity kleptoparasitism in old-growth longleaf pine (*Pinus palustris*) forests of southwestern Georgia by considering two competing hypotheses: 1) limited local snag availability influences the frequency of cavity usurpation, or 2) nesting in Red-cockaded Woodpecker cavities promotes higher nest success than nesting in snags. We used linear regression models to relate snag densities to occurrences of cavity usurpation and logistic-exposure models to measure combined nest success of two common cavity

competitors, Red-headed Woodpeckers (*Melanerpes erythrocephalus*) and Red-bellied Woodpeckers (*M. carolinus*), nesting in snags and usurped Red-cockaded Woodpecker cavities. Snag densities and kleptoparasitism occurrences were not related (2018: $r^2 = 0.109$, $F_{1,6} = 0.736$, $P = 0.424$; 2019: $r^2 = 0.062$, $F_{1,6} = 0.394$, $P = 0.553$), though more cavities were usurped where snags were limiting. The most supported model was cavity height. Nest success significantly improved ($Z = 2.84$, $P < 0.005$) in higher cavities, presumably due to increased predation rates at lower heights. Red-cockaded Woodpecker cavities were slightly lower than cavities in snags and daily nest success was lower in Red-cockaded Woodpecker (0.31) cavities than in snags (0.51), though the difference was not statistically significant. We conclude that 1) snag retention may alleviate competition, but high-quality Red-cockaded Woodpecker habitat will eventually host a surplus of cavities, and 2) Red-cockaded Woodpecker cavities may be potential ecological sinks for nesting avian kleptoparasites due to differential predation. This preliminary study warrants future work aimed at investigating potential costs and benefits to cavity kleptoparasitism.

Do similar foragers flock together? Non-breeding foraging behavior and its impact on mixed-species flocking associations in a subtropical region

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Mixed-species flocks are ubiquitous in forest bird communities, yet the extent to which positive (facilitative) or negative (competitive) interactions structure these assemblages has been a subject of debate. We described the fine-scale foraging ecology and used network analysis to quantify mixed-species flocking interactions of an insectivorous bird community in hardwood forests of north-central Florida. Our goal was to determine if similarly foraging species were more (facilitation hypothesis) or less (competition hypothesis) likely to associate in flocks, and if foraging ecology could explain intraspecific abundance patterns within flocks. We quantified attack maneuvers, foraging substrate, and foraging microhabitat of all 17 common insectivorous species in these forests and characterized the composition of 92 flocks encountered. Flocking was important in our community; 14 of 17 species joined more than 5% of flocks, and 10 species had flocking propensities of over 0.80.

Our results supported both hypothesized mechanisms structuring flock composition. Species had distinct, well-defined foraging niches during the non-breeding season, but foraging niche overlap among flocking species was greater than expected by chance. Consistent with the facilitation hypothesis, we found that similarly foraging species were significantly more likely to associate in flocks, a result driven by lower association strengths in large-bodied woodpeckers. We found no evidence of assortment by foraging behavior, however, likely because foraging behavior and substrate use showed strong niche partitioning at the fine scale within our community. Intraspecific abundance patterns were significantly linked to foraging substrate use, with live leaf use correlated with high within-flock abundance and relative abundance at study sites. Species that specialized on comparatively less-abundant substrates (tree trunks, epiphytes, dead leaves) joined flocks as singletons, showed lower relative abundance, and may exhibit non-breeding territoriality. Our results highlight the importance of foraging substrate use and mixed-species flocks in structuring the non-breeding ecology of migratory birds.

Urban greenspace is for the birds: nest box selection by Eastern Bluebirds on an urbanizing university campus

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Eastern Bluebirds (*Sialia sialis*) are secondary cavity nesters that underwent a population decline in the late 1900s due to a loss of natural nest cavities, and subsequent recovery due to an increase in artificial nest boxes. As urban development spreads, it is important to understand how to increase urban land use by native wildlife. Attractive nest sites for bluebirds typically include open grassy areas for foraging. In an urban context, we hypothesized that anthropogenic noise could significantly discourage nest box occupancy even if adequate grass is available. We monitored 78 identical nest boxes (Gilbertson style) spread across the University of Florida campus throughout the 2019 breeding season and compared features of occupied and unoccupied nest boxes. We calculated noise levels by taking three-minute samples of sound (in dBA) at four random times at each nest box and then taking the overall average decibel level of the samples. Additionally (based on bluebird nest box literature), we recorded box orientation,

percent grass (in a 300-m buffer), visibility from nest, distance to high perch, distance to fresh-water source, human activity level, distance to road, distance to nearest grassy open area, and distance to occupied building. To screen variables for analysis, we ran pairwise correlations followed by principal components analysis to combine, center, and scale collinear metrics. Finally, we used generalized linear models with binomial distribution and logit link function to assess the relationships between predictor variables and nest box occupancy. We found that nest boxes were significantly more likely to be occupied where percent grass was higher, noise was lower, and distance to buildings, roads, and water was greater. We conclude that within the urbanizing matrix, areas of quieter greenspace will be necessary to attract breeding populations; therefore, successful nesting habitat for the Eastern Bluebird can be integrated with greenspace planning.

Freshwater microplastic dissemination in retention ponds

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Urban retention ponds receive water input from rain and surface input from local areas. This study tests if microplastics, a product of chemical weathering and human pollution, rise in pollutant levels before and after a large rainfall event. Statistical quantification of microplastics in an environment usually takes place in marine environments, where chemical processes result in rapid dissimilation of human waste into the environment, and where the results are more extreme. However, few studies have been conducted on the total pollution of urban wastewater ditches, where plastics are more likely to have a total impact on the environment because of higher concentrations of waste and fluctuating water levels. This project focused on extracting microplastics from weekly water samples from the ponds, to determine if rapid rainwater input results in rapid pollution. We took four weeks of water samples from three separate ponds that shared a drainage basin, processed them with a dissecting microscope, and identified 350 plastics.

Ant presence on Red-cockaded Woodpecker (*Dryobates borealis*) selected trees in two southeastern forest types

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The Red-cockaded Woodpecker (*Leuconotopicus borealis*) is an endangered species native to the southeastern United States. We sought to create a species presence profile of the acrobat ant (*Crematogaster pinicola*), a species believed to be important as forage for the Red-cockaded Woodpecker. Two different, yet common, forest types that contain Red-cockaded Woodpeckers include old field and longleaf pine. We collected old field ant samples from Tall Timbers Research Station in north Florida with forests composed of loblolly (*Pinus taeda*) and shortleaf (*P. echinata*) pines with occasional longleaf pine (*P. palustris*). We collected longleaf ant samples from two different plantations located in south Georgia that were composed of predominantly longleaf pine. We baited trees foraged on by Red-cockaded Woodpeckers and a random control tree nearby with a cat food slurry. If Red-cockaded Woodpeckers prefer trees with more acrobat ants, then we expect to find a higher likelihood of acrobat ants on those trees that woodpeckers select. We also recorded the time Red-cockaded Woodpeckers spent on the trees relative to the presence of the acrobat ant as another metric of Red-cockaded Woodpecker preference. We counted all ant species on the bait and identified ants to the lowest possible taxonomic level. To date, we have observed the red imported fire ant (*Solenopsis invicta*), the acrobat ant, and four other ant species.

Linking bird community composition, diversity, and Northern Mockingbird song complexity along urbanization gradients in Florida

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Urbanization is one of the greatest threats to biodiversity around the world. By the end of this century, the proportion of urban land area is projected to increase by 185%, with the greatest expansions occurring in tropical ecosystems. Despite these current trends, only a few studies have investigated the impacts of urbanization on tropical species diversity. In addition, our understanding of the drivers and consequences of these changes in tropical diversity remains limited. Some evidence suggests that patterns of community structure along urbanization gradients are largely determined by the interaction between land use and species life-history traits, with species loss occurring predictably in species that share certain traits (environmental filtering). Changes in the composition of urban bird communities can also change the behaviors species use when they interact with other species. One such example is vocal mimicry, where species copy heterospecific vocalizations in their surrounding community. Previous studies have shown that vocal mimicry diversity and song complexity play an important role in male performance for multiple species. Because urbanization changes the species in the community available to mimic, urbanization may negatively affect the effectiveness of their communication. My research investigates the effects of urbanization on avian species diversity and life-history trait diversity in Florida, and vocal mimicry diversity and song complexity in the Northern Mockingbird. We predict that as urban land area and background noise levels increase, species richness, abundance, and life-history trait diversity will decline. We also predict that Northern Mockingbird vocal mimicry richness and song complexity will decline as a result of increased urbanization levels.

Florida Field Naturalist 49(1):30, 2021.

IN MEMORIAM: JOHN MURPHY, 1959–2020

John Michael Murphy was an inspired naturalist who made substantial contributions to avian conservation and the Florida Ornithological Society. He was born on November 10, 1959 in Tallahassee, Florida, and died at home in Franklin County, Florida on September 21, 2020 after a courageous two-year battle with cancer.

John grew up in Tallahassee and became interested in birds as a child through an aunt who took him on birding walks. He graduated from Florida State University and then had a 30-year career working for the Florida legislature. On retirement, he was able to further pursue his volunteer work with Florida Audubon, the U.S. Fish and Wildlife Service, and the Florida Division of Recreation and Parks, mostly tracking, banding, and maintaining data on nesting shorebirds along the north Florida coast from St. Marks to Cape San Blas. He was especially interested in the bird life of St. George Island in Franklin County.

He was a valuable member of the FOS, serving as Treasurer (2012-2014), on the Records Committee (2008-2014), and as a regional compiler for the Field Observations section of the *Florida Field Naturalist* from 2010 until his death.

John was also an active public servant in Franklin County and Alligator Point where he resided. He was a member of the Alligator Point Volunteer Fire Department and the Alligator Point Taxpayers Association. He also served on the Franklin County Planning and Zoning Committee.

Self-effacing, integrity personified, John was a good friend, a true gentleman, and a keen, sharp student of birds. Someone who knew him said that the world needed more John Murphys. It sure does.

I am grateful to Deanna Murphy and R. Todd Engstrom for their help with this memoriam.

—Robert L. Crawford
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HABITAT USE AND MOVEMENTS OF FERAL MALLARDS (*Anas platyrhynchos*) AND INVASIVE MUSCOVY DUCKS (*Cairina moschata*) IN TAMPA, FLORIDA

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Abstract.—Feral Mallards (*Anas platyrhynchos*) and invasive Muscovy Ducks (*Cairina moschata*) are common waterfowl in urban areas in many parts of the world, though relatively little is known about their habits and movements. This study quantified the fine-scale movements and habitat-use patterns of feral Mallards and Muscovy Ducks in an urban area of Tampa, Florida, from hour-long observational and tracking data. Mallards spent more time in water than Muscovy Ducks (41.0% vs. 9.1%), less time in grass (2.8% vs. 21.5%), and less time in urban areas (0.5% vs. 8.0%). Times spent by Mallards and Muscovy Ducks in tree-shrub (37.0% and 48.9%) and shoreline landscapes (18% and 12.5%) were statistically similar. Hourly distances traveled were similar between Mallards and Muscovy Ducks, averaging 250 m and 300 m, respectively. Mean hourly home ranges for Mallards were approximately 1,540 m^2, whereas those for Muscovy Ducks averaged about 900 m^2. The results suggest that feral Mallards and invasive Muscovy Ducks use similar wetlands and tree cover but differ in their use of other upland cover types. This work has implications for the management of native waterfowl, such as the Florida Mottled Duck (*Anas fulvigula fulvigula*), which also occupies urban areas and is threatened by interbreeding with feral Mallards.

Key words: habitat, home range, waterfowl, invasive species

Mallards (*Anas platyrhynchos*) and Muscovy Ducks (*Cairina moschata*) are common waterfowl in urban areas in many parts of the world, including Florida (Figley and VanDruff 1982, Johnson and Hawk 2009, Jarman et al. 2020). Wild Mallards are native migrants to Florida, although they are not a traditional part of the state's breeding fauna. However, feral Mallards—descended from locally released, domestically raised individuals—have established

non-migratory, breeding populations across the state (Fish 2011). Muscovy Ducks are non-native to Florida and considered invasive, with resident populations likewise established in many urban areas (Avery and Moulton 2007, Bielefeld et al. 2020). Potential threats from feral Mallards (hereafter Mallards) and Muscovy Ducks include disease transmission to native wildlife and domestic poultry (Wang et al. 2013, Elbestawy et al. 2019, Liu et al. 2020), water quality degradation (Liste et al. 2013, Nnaji 2014), and interbreeding with native waterfowl (Islam et al. 2013, Mo 2020). Hybridization with Mallards is a serious threat to the genetic identity of Florida Mottled Ducks (*Anas fulvigula fulvigula*), in particular. Although Mallards and Muscovy Ducks are hypothesized to also compete with native species for various resources (Varner et al. 2014, USFWS 2010), little is known about their home ranges, movements, and habitat-use patterns, especially where they are sympatric with one another and native ducks.

Feral Mallards in Florida occupy mostly aquatic cover in low- to high-intensity urban areas (Varner et al. 2014). In other locations, feral Mallards similarly have been shown to occupy urban ponds (Baratti et al. 2015, Meissner et al. 2015, Avilova 2016). Wild Mallards breed and winter in nearly every type of freshwater body but tend to avoid humans, built-up areas, dogs and cats, and farm premises (Sauter et al. 2012). Muscovy Ducks likewise occupy shallow urban ponds and wetlands (Julian and Galt 1980, Johnson and Hawk 2009), with a recent study documenting their usage of surrounding open lands for foraging, loafing, and preening activities (Downs et al. 2017). Another study reported the home range of an individual Muscovy Duck as 0.17 km^2 (Downs et al. 2014). However, no studies have systematically measured or compared movement and habitat-use patterns for feral Mallards and Muscovy Ducks. Our goal was to compare the fine-scale movements and habitat-use patterns of Mallards and Muscovy Ducks in an urban area of Tampa, Florida, using hour-long observational and tracking data. We used data for each species to 1) estimate the time spent in each land cover type, 2) calculate the hourly distance traveled, and 3) quantify their space-use patterns. We used our results to infer how the species may compete with one another or native waterfowl species, such as the Florida Mottled Duck.

Methods

Study area.—We conducted our study on the grounds of the University of South Florida Tampa campus located in the city of Tampa, Florida (28.06°N, 82.42°W). The Tampa campus is approximately 5 km^2 and contains waterfowl habitat used by Mallards and Muscovy Ducks that includes shallow wetlands and ponds, open grassy areas, and

sparse tree and shrub cover interspersed with other urban land (Anderson et al. 2017). The Tampa campus supports breeding populations of about 100 Muscovy Ducks (Cahanin 2017) and 65 feral Mallards (Z. J. Smith, University of South Florida, unpublished data).

Data collection.—We tracked Mallards and Muscovy Ducks by visual observation during June through November 2014 and April through July 2015. We used visual tracking methods because both species were accustomed to human activity and not disturbed by observer presence. Additionally, we found that visual tracking methods performed more accurately than global positioning system (GPS) tracking in pilot studies (Hyzer 2012, Downs et al. 2014), and the openness of the grounds allowed the ducks to be observed with few obstructions. We conducted all observations during daylight hours, approximately between 0700 and 2100, by naked eye or with the aid of binoculars. We observed ducks from a distance of 5 m to 15 m, as previous studies suggested they were not disturbed at this range (Downs et al. 2014). We began the tracking protocol by traveling to a pre-assigned quadrant of the campus and numbering any Mallards and Muscovy Ducks observed in a sequence from left to right. We selected one of those enumerated individuals for focal observation (Altmann 1974) using a random number generator. First, we recorded the date, time of observation, species, and sex. Second, we recorded the spatial coordinates of the duck to the nearest meter using a customized grid system overlaid with detailed satellite imagery. Third, we recorded the land cover type occupied by the duck according to five classes: water, shoreline (edge of wetland or pond), grass, tree or shrub cover, or urban (e.g., roads, parking lots, sidewalks). Finally, we tracked the focal duck for a one-hour duration and updated its spatial coordinates and land cover information every 30 seconds. If we could not observe a duck for any part of the tracking duration, we noted it as out of sight for those timestamps. We later imported all data into a geographic information system (GIS) for analysis. We collected a total of 188 hourly tracks (Mallards $n = 42$ and Muscovy Ducks $n = 146$) during the study (Fig. 1), divided roughly evenly between the first ($n = 96$) and second ($n = 92$) halves of the day.

Data analysis.—First, we summarized the percentages of points in each land cover type for each track. We used a factorial multivariate analysis of variance (MANOVA) to examine differences in the percentage of time spent in each land cover type between species and sexes of the same species (Davis and Smith 1998, Crook et al. 2009). We chose MANOVA because it is relatively robust to violations of multivariate normality and unequal variance. If the MANOVA detected differences between species or sexes ($P < 0.05$), then we used a univariate analysis of variance (ANOVA) to identify where those differences occurred.

Second, we summarized movement and space occupancy for each one-hour track. We estimated the distance traveled during each track by creating a space-time path. A space-time path, or trajectory, connects successive tracking points by a straight line and represents the minimum distance traveled during the tracking duration (Miller 2005). We calculated space-time path distances using a GIS and summarized distances by species and sex. We estimated hour-long space occupancy using time-geographic density estimation (TGDE) at three spatial scales: total occupied area, home range, and core area (Downs 2010). The TGDE method is a relatively new home range estimator that has been shown to perform accurately for frequently sampled tracking data (Downs et al. 2011). It blends traditional, statistical density estimation with trajectories to produce a continuous intensity surface from tracking data (Fig. 2). We applied TGDE to each track using a velocity multiplier of 1.05, as recommended by Downs et al. (2018), which generates a density surface based on the assumption that the individual could have traveled a distance no more than 5% of the calculated straight-line travel distance between each set of sample points. We calculated the total occupied areas, home ranges, and core areas from the 100%, 95%, and 50% isopleths, respectively, of the resulting TGDE surfaces.

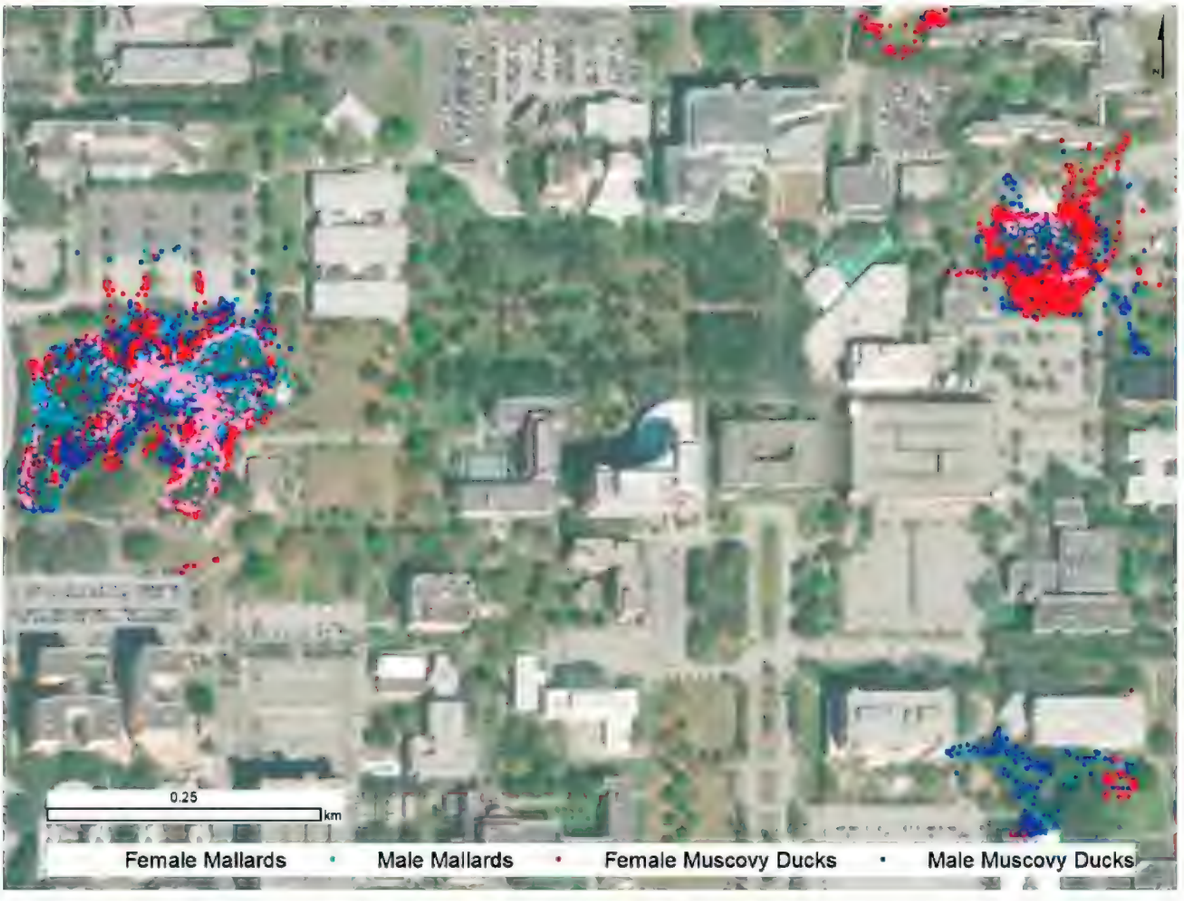

Figure 1. Tracking data collected for female Mallards (pink), male Mallards (light blue), female Muscovy Ducks (red), and male Muscovy Ducks (blue) at the University of South Florida Tampa campus, 2014–2015.

We used a Mann-Whitney U-test to detect differences in distance traveled and the three space-use measures between species and sexes within species. We chose the U-test because it is robust to data that are not normally distributed.

Results

We observed Mallards and Muscovy Ducks at two of the larger wetlands on campus but only Muscovy Ducks at two of the smaller wetlands (Fig. 1). Overall duck use of water, shoreline, grass, tree-shrub, and urban areas differed between species but not between sexes of the same species (Fig. 3). Mallards spent more time in water than Muscovy Ducks (41.0% vs. 9.1%), less time in grass (2.8% vs. 21.5%), and less time in urban areas (0.5% vs. 8.0%). Time spent by Mallards and Muscovy Ducks in tree-shrub (37.0% and 48.9%) and shorelines (18% and 12.5%) did not differ.

Given we found no differences between the sexes in spatial use parameters, we combined sexes within species for our estimates of distances traveled, hourly total occupied area, home range size, and core area size (Table 1). Mallards traveled approximately between 10 m and 1,400 m during hour-long observation periods, with a mean of 400 m and

Figure 2. Sample time-geographic density estimation surface generated for one-hour tracking data collected for a Muscovy Duck in Tampa, Florida, 2014–2015.

median of 380 m. Muscovy Ducks traveled between about 10 m and 1,500 m, with a mean of approximately 300 m and median of roughly 400 m. Median travel distances did not differ between species. Mallard total occupied areas, home ranges, and core areas were statistically larger than those for Muscovy Ducks (Table 1). Mallard hourly total occupied areas ranged from <1 m^2 to approximately 18,500 m^2 (approximate median ($\overline{x}$) = 950 m^2), whereas those for Muscovy Ducks were <1 m^2 to approximately 12,800 m^2 ($\overline{x}$ = 465 m^2). Mallard home ranges ($\overline{x}$ = 675 m^2) and core areas ($\overline{x}$ = 250 m^2) were approximately twice those of Muscovy Ducks ($\overline{x}$ = 330 m^2 and $\overline{x}$ = 120 m^2, respectively). Travel distances and all three measures of space occupancy were positively skewed for both Mallards and Muscovy Ducks, indicating that we observed periods of resting or relatively little movement more often than periods of frequent and longer movements.

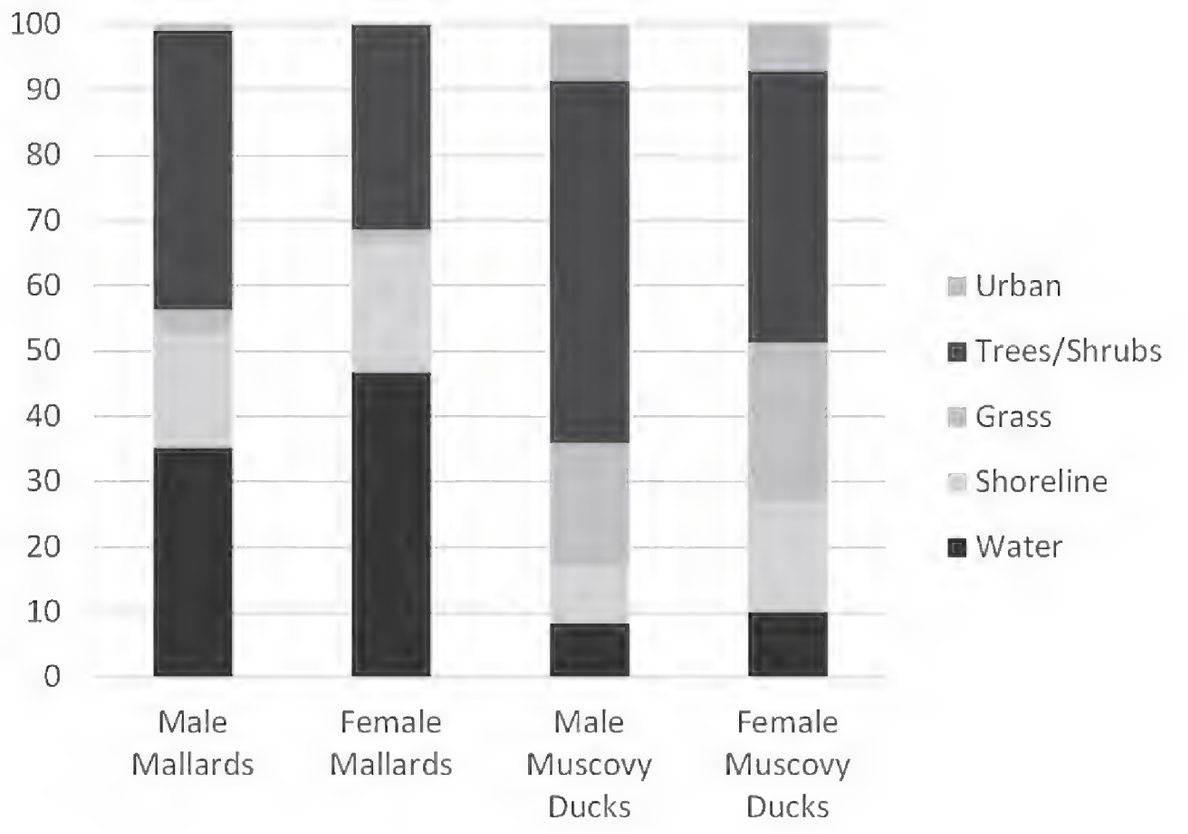

Figure 3. Proportions of time spent in five land cover types by male and female feral Mallards and Muscovy Ducks in Tampa, Florida, 2014–2015.

Discussion

We found that feral Mallards and Muscovy Ducks displayed different habitat- and space-use patterns while occupying similar wetlands. Mallards spent most of their time in water and adjacent shorelines, also using sparse tree-shrub cover. Muscovy Ducks spent relatively little time in water, spending more time in associated grass, tree-shrub, and urban cover types. Although the two species traveled similar distances during the hour-long tracks, Mallards occupied larger spatial areas than Muscovy Ducks. This implies that the movements of Mallards were spread out over larger areas, whereas Muscovy Ducks tended to move similar distances in more confined spaces. These patterns could be explained by differences in behavior and physiology between the two species. Muscovy Ducks forage in different land cover types by dabbling in water, grazing herbaceous vegetation, gleaning surface foods, probing the soil, and feeding from human handouts (Downs et al. 2017). This opportunistic foraging behavior, along with a limited ability to sustain long flights (Vidyadaran et al. 1991), could explain the relatively smaller hour-long home ranges of Muscovy Ducks. Similar studies of feral Mallards have not been conducted, but their wild counterparts make extensive use of wetlands and frequently

Table 1. Minimum, mean, standard deviation, median, and maximum distance traveled, total occupied area, home range, and core area for feral Mallard and Muscovy Ducks in Tampa, Florida, 2014–2015. All units are meters or square meters.

	Mallard (n = 42)				Muscovy (n = 146)			
	Distance	Total area	Home range	Core area	Distance	Total area	Home range	Core area
Minimum	10.8	0.1	0.1	0.0	11.8	0.1	0.1	0.0
Mean	404.4	2,183.9	1,541.9	567.2	311.9	1,303.7	910.1	332.7
SD	297.9	3,313.5	2,342.7	891.8	252.5	2,006.3	1,404.0	515.8
Median	380.9	950.0	675.3	248.2	402.8	464.2	328.9	118.4
Maximum	1,382.3	18,514.4	13,107.8	5,109.4	1,516.3	12,811.3	9,437.3	3,626.0

feed by dabbling (Johnson and Rohwer 2000), though they also will opportunistically forage on land by grazing, plucking seeds, or probing for roots (Kaufman 2001, Nummi 2011). Mallards have high flight capacities (Dial et al. 2012), which could also explain their larger hour-long home ranges as compared to Muscovy Ducks.

Additionally, spatial overlap between the movements of Mallards and Muscovy Ducks suggests that there could be competition between them for aquatic resources. This may be most notable during night because both species roost on shorelines of wetlands (Cahanin 2017). Though competition between the two populations is not a conservation concern, as they are both feral, understanding how they might affect or compete with native waterfowl is a management priority. The Florida Mottled Duck is a native species of particular concern it is non-migratory and up to half of its population occupies wetlands in urban areas (Bielefeld et al. 2010). Urban Florida Mottled Ducks occupy wetlands, including artificial impoundments, and use adjacent urban and upland land cover types (Varner et al. 2013, 2014) similar to those identified as used by feral Mallards and Muscovy Ducks in this study. This suggests there is potential for habitat competition and interaction between Florida Mottled Ducks and both feral species. Not only is competition for resources a potential concern, but hybridization with feral Mallards has been well established as a threat to the genetic identity of the Florida Mottled Duck (Seyoum et al. 2012, Bielefeld et al. 2020). Muscovy Ducks can hybridize with dabbling ducks, such as Mallards and Mottled Ducks, though their offspring are thought to be infertile and are less of a concern (Sercy et al. 1996). Likewise, disease transmission risks may be increased where the populations are sympatric because Mottled Ducks, Mallards, and Muscovy Ducks are all susceptible to avian flu (Ferro et al. 2008, Capua and Mutinelli 2008). The findings that feral Mallards and Muscovy Ducks use similar habitats to Florida Mottled Ducks has implications for their management. Aggressive removal of feral ducks in habitat occupied by Florida Mottled Ducks is an option, although their large numbers and widespread distribution may present a challenge. Ford (2017) suggests the best strategy for protecting native waterfowl like the Florida Mottled Duck is to conserve and restore natural wetlands, especially those in coastal and areas, to minimize the movements of native species into urban areas occupied by feral and invasive species. Because Florida Mottled Ducks move little between rural and urban areas, habitat conservation in rural, coastal areas may be the most effective conservation strategy (Varner et al. 2014).

Though this study characterized the habitats and movements of two feral species, there are some limitations. The observational data were limited to tracking durations of one hour, though the sampling

interval was frequent at 30 seconds. We selected this sampling protocol to understand the fine-scale similarities and differences between the habitat-use and movement patterns of the two species. Because the habitat-use patterns for the Muscovy Ducks in this study were similar to those reported by Downs et al. (2017) from detailed time-activity budgets, this suggests the collected tracking data are representative over short durations. However, the reported values for total areas, home ranges, and core areas should be considered only in the context of one-hour time periods and not extrapolated to infer daily or seasonal measurements because they may potentially be much larger. Additionally, we collected our data during the wet season of both years. Future work might aim to collect and analyze tracking data for the species at broader spatial and temporal scales, and during the dry season, because improved knowledge of their movements may lend insights into their impacts upon native waterfowl.

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CITY BIRDS: AVIAN ABUNDANCE AND DIVERSITY IN JACKSONVILLE, FLORIDA

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Abstract.—Urbanization is rapidly changing the landscape in which birds live. Habitat alteration and degradation and changes in food resources are affecting the distribution and abundance of birds in urban areas. Jacksonville is the largest city in Florida and is undergoing considerable growth. We investigated how urbanization might affect avian abundance and diversity at four sites in Jacksonville, Florida. For all sites we calculated land cover percentages (tree cover, ground cover, impervious surfaces, water, and shrubs) at three spatial scales (50-m, 100-m, and 1,000-m radius) to encompass the area sampled for birds and the larger urban matrix. We conducted point counts and compared counts of species and total individuals across all sites. Additionally, we examined diversity across all sites with the Shannon-Wiener diversity index and rank abundance curves. We examined eBird data to assess both bird species richness and urban bird community similarity. The most urbanized site, with the highest percent impervious surface, located in downtown Jacksonville had the lowest bird diversity; contrary to predictions this site also had the lowest abundance of birds. The other sites, including a city park, a college campus, and a national park had tree cover ranging from 43% to 100%, relatively high diversity, and community similarity, which indicated the importance of green space for birds living in the city. Data from eBird support our findings and larger green spaces, such as large forested parks within cities, may be vital for maintaining urban bird diversity.

Key words: urbanization, avian diversity, green spaces

Urbanization is transforming landscapes throughout the world, rapidly altering abiotic and biotic factors in the environment (Silva et al. 2015, Ibáñez-Álamo et al. 2017). Habitat loss and fragmentation, increased noise and light levels, and shifting food resources and predators are affecting biological communities, including many species of birds (Fernández-Juricic and Jokimäki 2001, Sandström et al. 2006, Grimm et al. 2008, Hagen et al. 2017). Approximately a fifth of known bird species are found in urban areas, making birds a model system to gather data on the effects of urbanization (Lepczyk et al. 2017).

Studies have documented decreases in avian diversity in urban areas but an increase in population density (Chace and Walsh 2006, Sandström et al. 2006, Loss et al. 2009, Shochat et al. 2010b). In urban areas the diversity and density of birds is related to food, nest site availability, and the type and amount of green spaces (i.e., natural patches versus man-made parks; Sandström et al. 2006, Fuller at al. 2008). For example, Northern Cardinal (*Cardinalis cardinalis*) population densities are four times higher in urban forests where food abundance is 2.6 times higher than rural forests (Shochat et al. 2010a). Even though food is abundant and predictable in urban environments, only certain species thrive in cities (Sandström et al. 2006, Møller 2008, Møller 2009).

Some avian species are considered urban adapted; they have higher tolerance levels to urbanization and often show behavioral changes in foraging and reproduction (Bonier et al. 2007, Møller 2008, Møller 2009, Shochat et al. 2010b, Aronson et al. 2014). Examples of urban-adapted birds include House Sparrows (*Passer domesticus*), Rock Pigeons (*Columba livia*), Barn Swallows (*Hirundo rustica*), and European Starlings (*Sturnus vulgaris*). These four species are found in over 80% of cities across the globe (Sandström et al. 2006, Aronson et al. 2014, Lepczyk et al. 2017). Although there are some urban-adapted birds, the majority of avian species seem to avoid urban cores (Greggor et al. 2016, Hagen et al. 2017, Ibáñez-Álamo et al. 2017).

Urban areas have the potential to support native avian biodiversity, but this requires conservation of green spaces and controlling the amount of urban development (Aronson et al. 2014). Urban areas consist of anthropogenic structures and impervious surfaces. They also have green spaces and natural parks scattered throughout, with increasing tree coverage as you move away from the urban core (American Forests 2005, Sandström et al. 2006, Loss et al. 2009, Aronson et al. 2014). By keeping conservation in mind and planning for natural green spaces, cities might be able to better support local avian biodiversity (Gaston et al. 2007, Aronson et al. 2017).

The majority of Florida's nearly 21 million people live in or near cities (Bureau of Economic and Business Research 2017) and rapidly growing populations in Florida have led to increased amounts of developed urban areas (Reynolds 2001). We investigated how urbanization in Florida's largest city, Jacksonville, is affecting bird abundance and diversity. We compared avian abundance, species richness, and diversity at four locations that varied in their levels of urbanization in Jacksonville, Florida. The four locations ranged from the urban core of downtown Jacksonville to residential areas and a national park. We predicted that highly urbanized areas, with more impervious surface area and less tree coverage, would have a higher abundance of birds but would be less diverse.

Methods

Study sites.—We picked four locations that varied in urbanization in Jacksonville to collect data on bird species diversity and abundance: the Duval County Courthouse, Jacksonville University, Boone Park, and Theodore Roosevelt Area (Fig. 1). The Duval County Courthouse is located in the center of downtown Jacksonville. Jacksonville University (JU) is in a residential area along the St. Johns River. The campus sits on 108.5 ha and consists of buildings, athletic fields, ponds, and green spaces. Boone Park is a 11.3-ha city park located in a residential area. The Theodore Roosevelt Area is a 242.8-ha portion of the National Park Service's Timucuan Ecological and Historic Preserve. The Theodore Roosevelt Area is a protected area consisting of hammock forests and wetlands, but it is surrounded by residential properties.

Bird data.—We gathered data for a year, from January 2017 to January 2018. At each location we conducted point counts to collect data on bird species diversity and abundance. Point count location was the same spot within each site for each visit. We visited each site a minimum of seven times total throughout the year (Duval County Courthouse = 7 visits, JU = 9 visits, Boone Park = 8 visits, and Theodore Roosevelt Area = 9 visits) and we visited each site at least once for each season (fall, winter, spring, and summer). We did not conduct more than one point count at each site per day. One observer (JMCS) conducted 20-minute point counts within a 20-m radius between the hours of 0700 and 1100. We identified bird species visually and by sound. We did not include bird species that were flying overhead (not using the site) in the count. Because our sampling effort was low, we examined eBird data gauge diversity and community similarity at each location (eBird 2012). We searched each study location, in the hotspot

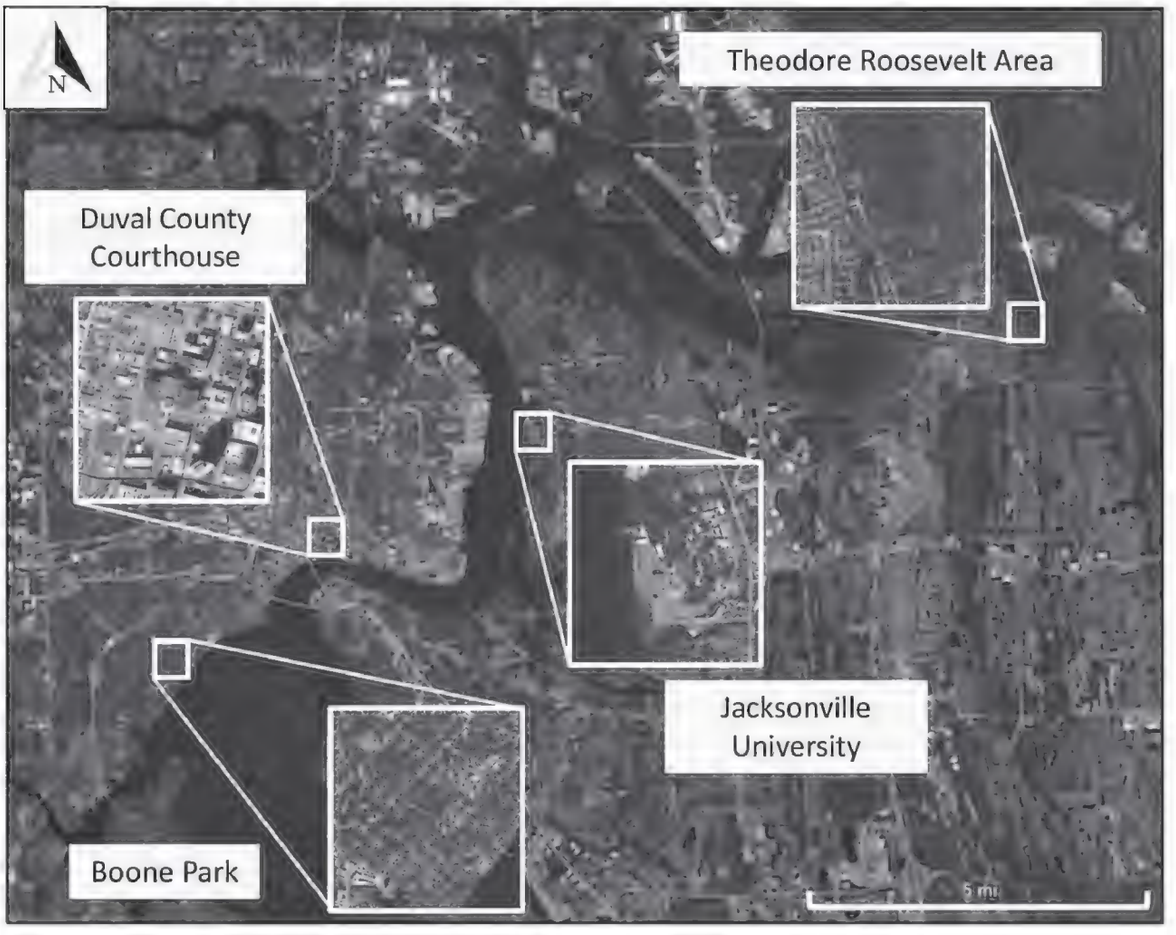

Figure 1. Bird survey sites in Jacksonville, Florida, 2017–2018.

search window on the eBird website. All sites, except for the Duval County Courthouse were listed as hotspots. For the sites listed as hotspots, we recorded the total number of species reported year-round and in all years. We verified the total species number for each site and gathered community composition data from the location checklists available from eBird. For the Duval County Courthouse, we chose the closest hotspot nearby, which was the Jacksonville Landing approximately 0.8 km from the courthouse along the St. Johns River. Although the proximity to the water may have increased richness, this was the nearest downtown hotspot location with similar infrastructure for which we could acquire bird numbers.

Data analyses.—To examine habitat differences between study sites and to rank level of urbanization across sites we defined an area of influence for each study site at three different spatial scales, representing a radius of 50 m, 100 m, and 1,000 m from the location of point counts. With these three spatial scales, we were able to capture habitat features at the scale of bird sampling and the larger habitat matrix surrounding the sampling points; habitat features, especially tree cover, are important for inside patches and the surrounding matrix (Taylor et al. 2016). For buffer size at each survey site, we measured the following parameters: percent tree cover, percent shrubs and bushes, percent lawns, percent impervious surface, and percent water. We used free online software developed by i-Tree Cooperative (USDA Forest Service, Davey Tree Expert Co., National Arbor Day Foundation, Society of Municipal Arborists, International Society of Arboriculture and Casey Trees) to analyze the percent vegetative cover for each selected area (i-Tree Canopy v6.1). The assessment method involves using Google Maps aerial photography at random points within the selected area. The sampling zooms to each point so the user can then choose from the pre-defined land cover types for that point. For the smaller spatial scales (50 m and 100 m), we used 499–500 sampling points per site and for the largest scale (1,000 m), we used 499–1,029 sampling points to maximize accuracy. We then used this land cover data across all three spatial scales to rank the sites from most urbanized (highest percentage of impervious surface, least green space) to least urbanized.

For each site, we counted the total number of birds present (bird abundance) and number of different species (bird diversity). Count data of birds and other organisms tends to be skewed and does not fit a normal distribution; therefore, we used a negative binomial test (PROC GENMOD) in SAS (SAS Institute, Cary, NC, USA) to compare bird counts (White and Bennetts 1996, Seavy et al. 2005, Stroup 2014) from the total number of birds obtained from point-count surveys at each site to assess differences in abundance among sites. To compare richness between sites, we compared counts of the number of species at each location from the point-count survey data. We also calculated the Shannon-Weiner index and Pielou's evenness index, and constructed rank abundance curves for each site using the point count survey data. Finally, we used the more robust eBird data set to examine species richness and assess community similarity across all sites using Jaccard's index.

Results

We found the same trend in tree cover and impervious surfaces at all three spatial scales. Theodore Roosevelt Park had the highest tree cover of all the sites ranging from 75–100%, followed by Boone Park (50–80%), and JU (43–58%). The Duval County Courthouse had the lowest tree cover of all the sites (3–12%). In terms of impervious surfaces, the Duval County Courthouse was highest (54–74%), followed by Boone Park (16–35%) and JU (14–23%). Theodore Roosevelt Park

had the lowest amount of impervious surface, ranging from 0–10% and at both the 50-m and 100-m radius, tree cover was 100%. Ground cover varied depending on the spatial scale examined. Although percent water cover was near or at 0% for the 50-m or 100-m scale, within the larger matrix of 1,000 m, water was a component of cover, especially for JU and the Duval County Courthouse, which were closer to the river (range = 11–18%). Shrub and bushes were similar across all sites with a range from 0–4% (Table 1).

We identified 32 species during point counts (Table 2). The total number of birds counted differed among sites ($\chi^2 = 8.87$, $P = 0.031$). Bird abundance was highest at JU, followed by Theodore Roosevelt, Boone Park, and the Courthouse (Table 3). The number of species counted at each location also differed ($\chi^2 = 19.23$, $P \leq 0.001$). In addition to having the most birds, JU also had high species richness (Table 3). The Theodore Roosevelt Area had the highest Shannon Wiener index and greatest evenness, but JU and Boone Park values also indicated relatively diverse and even communities. The Courthouse had the fewest species, and lowest diversity and evenness. The rank abundance curves also indicated low diversity at the Courthouse in comparison to the other sites, which had greater species richness and evenness (Fig. 2). We counted fewer species than were documented on eBird at our sites. The eBird data indicated highest richness at Theodore Roosevelt (192 species), followed by JU (111 species), Boone Park (78 species), and the Courthouse (66 species). The Duval County Courthouse bird community was the most dissimilar to the other areas and JU avifaunal community was very similar to both Boone Park and the Theodore Roosevelt Area (Table 4).

Discussion

Urban areas tend to have higher population densities but lower diversity in comparison to less urbanized areas (Lepczyk et al. 2017). Our data for Jacksonville support these trends; we found significant differences in avian abundance and richness between sites, with more urbanized areas lacking diversity. Although all our sites were located within the urban matrix of Jacksonville, the degree of urbanization (i.e., percent impervious surface, percent tree coverage) within and surrounding each site varied. Lack of tree coverage and natural green space at the Duval County Courthouse likely reduced avian biodiversity in the area. The courthouse had the lowest level of tree cover and the highest amount of impervious surface of all sites and as predicted the courthouse had the least diverse bird community. Other studies have shown that increasing impervious surface with urbanization decreases bird diversity (Evans et al. 2018, Carvajal-Castro et al. 2019).

Table 1. Land cover at three spatial scales (50-m, 100-m, 1,000-m radius) across the four study sites in Jacksonville, Florida, 2017–2018 (n = number of sample points at each site to determine land cover types).

		50-m radius			100-m radius			1,000-m radius		
	Cover class	n	% cover	SE	n	% cover	SE	n	% cover	SE
Theodore Roosevelt										
	Trees	499	100.00	0.00	500	100.00	0.00	375	75.00	1.94
	Ground cover	0	0.00	0.00	0	0.00	0.00	45	9.00	1.28
	Impervious	0	0.00	0.00	0	0.00	0.00	50	10.00	1.34
	Water	0	0.00	0.00	0	0.00	0.00	10	2.00	0.63
	Shrubs and bushes	0	0.00	0.00	0	0.00	0.00	20	4.00	0.88
Boone Park										
	Trees	404	81.00	1.76	388	77.60	1.86	512	50.00	1.56
	Ground cover	6	1.20	0.49	29	5.80	1.05	132	13.00	1.04
	Impervious	89	17.80	1.71	79	15.80	1.63	360	35.00	1.49
	Water	0	0.00	0.00	0	0.00	0.00	3	0.30	0.17
	Shrubs and bushes	0	0.00	0.00	4	0.80	0.40	22	2.10	0.45
Jacksonville University										
	Trees	266	53.20	2.23	291	58.20	2.21	215	43.00	2.21
	Ground cover	143	28.60	2.02	94	18.80	1.75	73	15.00	1.58
	Impervious	70	14.00	1.55	104	20.80	1.82	117	23.40	1.89
	Water	0	0.00	0.00	0	0.00	0.00	88	18.00	1.70
	Shrubs and bushes	21	4.20	0.90	11	2.20	0.66	7	1.40	0.53
Duval County Courthouse										
	Trees	32	6.40	1.09	18	3.59	0.83	121	12.00	1.03
	Ground cover	196	39.20	2.18	113	22.60	1.87	111	11.00	0.99
	Impervious	271	54.20	2.23	370	73.90	1.96	649	64.90	1.51
	Water	0	0.00	0.00	0	0.00	0.00	108	11.00	0.98
	Shrubs and bushes	1	0.20	0.20	0	0.00	0.00	11	1.10	0.33

Table 2. Bird species counted at four locations in Jacksonville, Florida, 2017–2018.

Common name	Scientific name	Location[a]
Canada Goose	*Branta canadensis*	JU
Mourning Dove	*Zenaida macroura*	TRA, BP, JU, DCC
Laughing Gull	*Leucophaeus atricilla*	JU
Unknown gull spp.	N/A	DCC, JU
Great Blue Heron	*Ardea herodias*	TRA
Barred Owl	*Strix varia*	TRA
Red-bellied Woodpecker	*Melanerpes carolinus*	TRA, BP, JU
Downy Woodpecker	*Dryobates pubescens*	TRA, BP, JU
Hairy Woodpecker	*Dryobates villosus*	TRA, BP, JU
Pileated Woodpecker	*Dryocopus pileatus*	TRA, BP, JU
Unknown woodpecker spp.	N/A	TRA, JU
Great Crested Flycatcher	*Myiarchus crinitus*	BP
White-eyed Vireo	*Vireo griseus*	TRA
Red-eyed Vireo	*Vireo olivaceus*	TRA
Blue Jay	*Cyanocitta cristata*	BP, JU
American Crow	*Corvus brachyrhynchos*	BP, JU
Fish Crow	*Corvus ossifragus*	BP, JU
Unknown crow spp.	N/A	TRA, DCC
Tree Swallow	*Tachycineta bicolor*	JU
Carolina Chickadee	*Poecile carolinensis*	JU
Tufted Titmouse	*Baeolophus bicolor*	TRA, BP, JU
Carolina Wren	*Thryothorus ludovicianus*	TRA, BP, JU, DCC
Ruby-crowned Kinglet	*Regulus calendula*	TRA
Eastern Bluebird	*Sialia sialis*	BP
Gray Catbird	*Dumetella carolinensis*	TRA
Brown Thrasher	*Toxostoma rufum*	JU
Northern Mockingbird	*Mimus polygottos*	BP, JU, DCC
European Starling	*Sturnus vulgaris*	BP, DCC
House Sparrow	*Passer domesticus*	DCC
House Finch	*Haemorhous mexicanus*	JU
Boat-tailed Grackle	*Quiscalus major*	DCC
Northern Parula	*Setophaga americana*	TRA, BP, JU
Black-throated Blue Warbler	*Setophaga caerulescens*	TRA
Yellow-rumped Warbler	*Setophaga coronata*	TRA, JU
Northern Cardinal	*Cardinalis cardinalis*	TRA, BP, JU

[a]Theodore Roosevelt Area (TRA), Boone Park (BP), Jacksonville University (JU), Duval County Courthouse (DCC).

The Theodore Roosevelt Area, JU, and Boone Park had greater percentages of tree cover and less impervious surface. The Theodore Roosevelt Area is surrounded by a residential area, but its dense tree coverage, larger size, and variation in land cover types (marshes, scrub vegetation, forests) could increase bird diversity. Other studies have also documented the importance of trees in increasing bird diversity (Fontana et al. 2011, Barth et al. 2015, Threlfall et al. 2016).

Table 3. Urban bird abundance, diversity, and evenness in Jacksonville, Florida, 2017–2018.

Site	Bird abundance	Species richness	Shannon Wiener index (H)	Evenness (J)
Theodore Roosevelt Area	141	20	2.42	0.81
Boone Park	120	18	2.23	0.77
Jacksonville University	191	24	2.32	0.73
Duval County Courthouse	92	9	1.20	0.52

Jacksonville University's large size, considerable amount of tree coverage and shrubs, and location on the St. Johns River could allow for more species to be present even though JU is next to downtown. The bird community at JU was also more similar to other sites and JU shared 80% of the birds found at the most diverse area, Theodore Roosevelt. Boone Park is located within a residential neighborhood, but both the park and the neighborhood have considerable tree coverage.

In addition to tree cover, area is an important component in determining avian abundance in urban parks (Melles et al. 2003, Strohbach et al. 2013, Schutz and Schulze, 2015). The lower species richness at Boone Park in comparison to the Theodore Roosevelt Area and JU may be due to the small size of the park. Despite its smaller

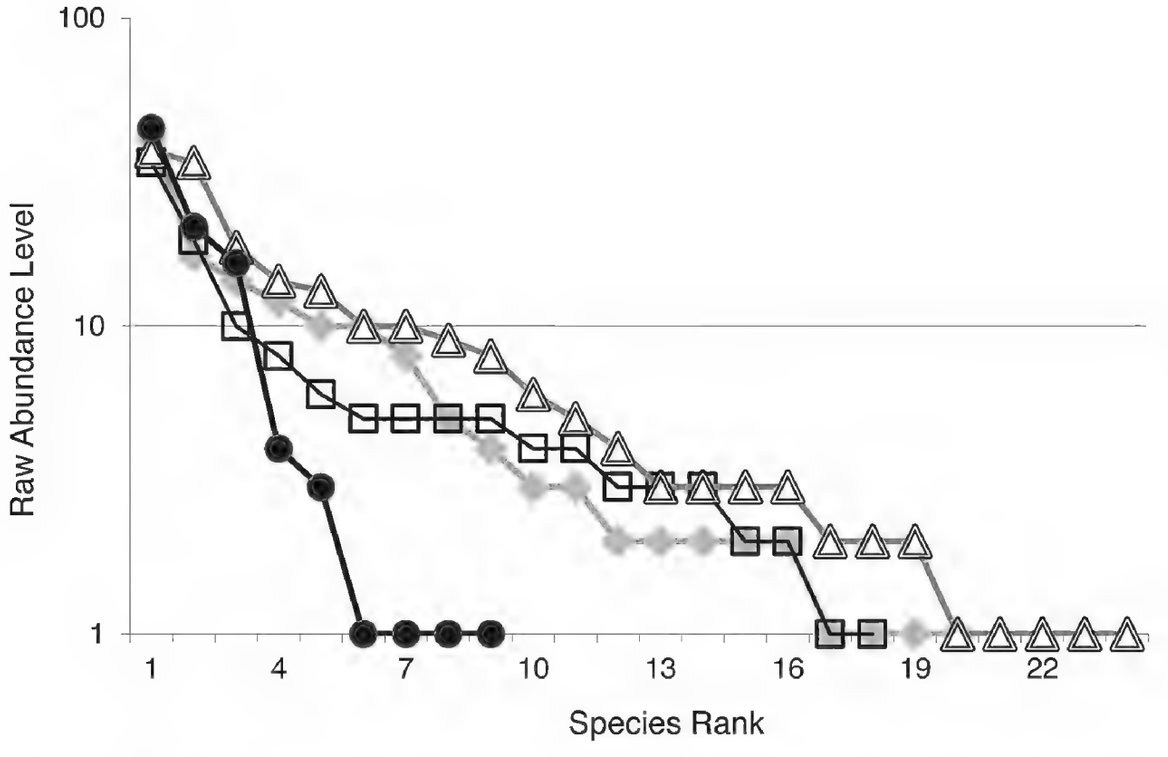

Figure 2. Rank abundance curves of avian diversity at four sites (Diamond = Theodore Roosevelt Area, Square = Boone Park, Triangle = Jacksonville University, Circle = Duval County Courthouse) in Jacksonville, Florida, 2017–2018.

Table 4. Jaccard similarity coefficients for four urban bird communities in Jacksonville, Florida, 2017–2018.

	Boone Park	Jacksonville University	Jacksonville Landing
Theodore Roosevelt	0.69	0.80	0.64
Boone Park		0.77	0.68
Jacksonville University			0.65

size, the Shannon Wiener diversity index was higher for Boone Park than JU. Boone Park also had close to 75% of the species found at JU and the Theodore Roosevelt Area. Area may also account for why we did not see greater abundance of birds at the Duval County Courthouse. The results of this study could have also been affected by a relatively small sampling effort; however, the eBird data on number of species at each location matched the trends seen in our data in terms of species richness. The eBird data also aligned with tree cover (i.e., the areas with the highest amount of tree cover across all spatial scales had the most species).

Tree coverage, green spaces, and natural patches in cities affect the variety and number of niches for birds, so identifying changes within avian communities may help us predict how avian species and other animals are responding to urbanization (Chace and Walsh 2006, Hunter and Hunter 2008, Aronson et al. 2014, Ibáñez-Álamo et al. 2017). By adding natural patches and corridors, more habitat space and resources could be provided for avian species in urban areas (Beninde et al. 2015). Cities typically provide different resources of food, including anthropogenic food that typically lacks nutrients, artificial feeders (songbird and hummingbird), and natural resources provided by natural patches (Hagen et al. 2017). The predictability and proportion of these food resources available could greatly affect the avian biodiversity, especially if the majority of urban areas have artificial feeders and nutrient-lacking anthropogenic food (Shochat et al. 2010a, Hagen at al. 2017). The Duval County Courthouse lacks large numbers of trees and natural green space that could provide foraging opportunities for birds. We observed avian species near the courthouse foraging on anthropogenic food (e.g., bread from a hot dog cart), whereas the species at the other three sites would forage on more natural food items, including invertebrates. Birds rely on a variety of plant and invertebrate species for resources that are crucial to their survival; a low diversity of avian species in a patch could be correlated to low amounts of plant and invertebrate species (Silva et al. 2015). This may be especially true for warbler species and we did not document any at the courthouse; instead, the courthouse was the only location where we found the urban-adapted House Sparrow.

As long as green spaces and forested patches remain throughout cities, there is hope for sustaining avian biodiversity (Aronson et al. 2014). Being able to plan purposeful green spaces and urban forests could be a solution to increase urban avian biodiversity (Threlfall et al. 2016, Aronson et al. 2017). A recent study in Beijing, China, noted an increase in forest cover in the city may account for an increase in bird species richness; from 1987 to 2014, the city gained a net total of 81 bird species (Pei et al. 2018). Monitoring of Jacksonville's green spaces and forested areas can help us understand how fast urbanizing is occurring, and how distinct areas and land cover types within the city might be changing over time (American Forests 2005). An understanding of how urbanization in Jacksonville and other cities affects avian biodiversity will be important for gaining insight on how other organisms and overall environmental health is being affected as cities rapidly change.

Acknowledgments

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SATELLITE TELEMETRY REVEALS HABITAT SELECTION AND MOVEMENT PATTERNS OF AN AMERICAN FLAMINGO IN FLORIDA BAY

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Abstract.—American Flamingos are among the most iconic—and rarest—of Florida's native birds. Though historical large flocks were decimated by hunting, small groups of flamingos of unknown origin persist in Florida today. Here, we report a satellite telemetry study of an American Flamingo in Florida Bay, Florida, USA, over a 22-month period. The flamingo used an extensive tidal flat at Snake Bight (Everglades National Park) and numerous mangrove-fringed mudflats inside keys within Florida Bay. Movement patterns varied by time of day and time of year and indicated that movements were tracking hydroperiod and prey availability across a landscape mosaic of spatially and temporally variable resources. We used a community science platform (eBird) to assess reporting probability for this flamingo; reporting probability was very low despite the conspicuous nature of the species. Though our telemetry data are from a single individual, this study represents the first empirical field study of any wild flamingo in Florida. Our findings on habitat selection and movement will be foundational to developing evidence-based conservation strategies for flamingos.

Key words: American Flamingo, Florida Bay, habitat selection, satellite tracking, residency

American Flamingos (*Phoenicopterus ruber*) are among the most iconic of Florida's birds. In the nineteenth century, naturalists described flocks of hundreds or thousands of flamingos in Florida Bay and the Florida Keys (Audubon 1839, Wurdemann 1860, McCall 1868, Scott 1890, Ingraham 1893, Howe 1902) and flamingos appear to have nested within Florida (Sprunt 1937, Whitfield et al. 2018). However,

by the end of the nineteenth century, flamingo populations throughout the Caribbean and the Bahamas were decimated by hunting (Allen 1956, Rooth 1965, Sprunt 1975, Baldassarre and Arengo 2000) and the Florida population was believed to be extirpated (Scott 1890, Allen 1956, Whitfield et al. 2018). Through the twentieth century, captive flamingos in public animal attractions and private collections likely outnumbered their wild counterparts in Florida and most ornithologists considered flamingo observations in natural areas to be escapees from captive colonies (Bailey 1932; Allen 1954, 1956; Stevenson and Anderson 1994).

Several lines of evidence, however, suggest that Florida's contemporary flamingos are not escapees of captive origin and may instead represent early signs of a recolonizing population. Flamingo populations in the Bahamas and much of the Caribbean rose sharply through the late twentieth century (Allen 1956, Sprunt 1975, Baldassarre and Arengo 2000) and flamingos have started to recolonize nesting areas from which they were historically extirpated (Wiley 1979, Paulino et al. 2011, Sanz D'Angelo 2020). Expanding flamingo populations across the Caribbean are consistent with an increasing number of flamingo sightings in Florida (Whitfield et al. 2018), though to our knowledge recent nesting has not resumed in Florida. On three occasions, flamingos that were banded as chicks in the Yucatan region of Mexico have had confirmed sightings within Florida, illustrating natural connectivity between Florida and other nesting areas (Galvez et al. 2016, Whitfield et al. 2018, Welsh and White 2019). Occurrence records in the Florida panhandle and northeastern Florida also suggest that flamingo sightings there are the result of natural displacement by tropical storms rather than escapees (McNair and Gore 1998, Pranty and Basili 2007).

Most flamingos within Florida are sighted in Florida Bay (Whitfield et al. 2018), a shallow estuary between the southern tip of the Florida peninsula and the Florida Keys (Fourqurean and Robblee 1999, Wingard et al. 2019). Florida Bay represents a mosaic of geomorphological landscape features that vary in water depth, salinity, and hydroperiod—all of which are likely to influence suitability to flamingos for foraging or nesting sites. Much of Florida Bay is remote, inaccessible, or has restricted access; therefore, flamingos that do appear in the area may go unreported.

In the absence of monitoring programs for flamingos, the only data for evaluating population size and population trends are sourced from community science programs (historically called citizen science programs). One well-known example of a community science platform, eBird, has leveraged observational information from recreational birdwatching to create global-scale georeferenced and timestamped

observation data for bird species (Sullivan et al. 2009, 2017). Whitfield et al. (2018) recently used data from eBird and other community science platforms to identify trends in flamingo abundance in Florida over several decades. However, community science programs vary in spatial, temporal, and taxonomic coverage resulting in biases in reporting probability (Bird et al. 2014), although large and conspicuously colored species are more likely to be reported than nondescript species (Caley et al. 2020). Because flamingos are both large and brightly colored, reports from eBird may accurately track abundance and distribution. Ultimately, it is difficult to interpret the efficacy of community science programs to reflect actual trends in flamingo abundance without empirical validation.

Satellite telemetry may provide useful insights into the behavior and ecology of American Flamingos in Florida and help to clarify reporting probabilities from community science programs. For other flamingo species, satellite telemetry has revealed long-distance movements that elucidate population connectivity (Amat et al. 2005) and illustrate linkages between foraging and nesting areas (McCulloch et al. 2003). Unlike banding and resighting programs that provide occasional evidence for movements (Sprunt 1975, Galvez et al. 2016) or traditional radio-telemetry, which is labor intensive and not feasible at international spatial scales, satellite telemetry has the advantage of providing multiple locality observations at regular time intervals from almost any location on the planet. Satellite telemetry has been used successfully in other species of flamingos (McCulloch et al. 2003, Amat et al. 2005, Balkiz et al. 2010) but, to our knowledge, has not been reported from American Flamingos.

Here, we report the results of satellite telemetry data for an American Flamingo in Florida Bay. We used these data to seek preliminary answers to several fundamental questions regarding the biology of flamingos in Florida. How are flamingos in Florida connected to nesting or foraging areas outside the state? What landscape features are important habitat for flamingos in Florida Bay? What are the daily and seasonal movement patterns of flamingos? How probable is the reporting of flamingos via the eBird community science project? The answers to these questions should be foundational to developing evidence-based wildlife policies and management practices for one of Florida's most iconic species.

Methods

Capture, rehabilitation, and release.—Our study flamingo (hereafter, Conchy) first appeared at the Naval Air Station (NAS) Key West at Boca Chica Key, Florida, USA (Fig. 1) on 1 September 2015 with two other flamingos foraging in shallow lagoons adjacent to the runways (Gonzalez 2015). The two other flamingos departed, leaving Conchy

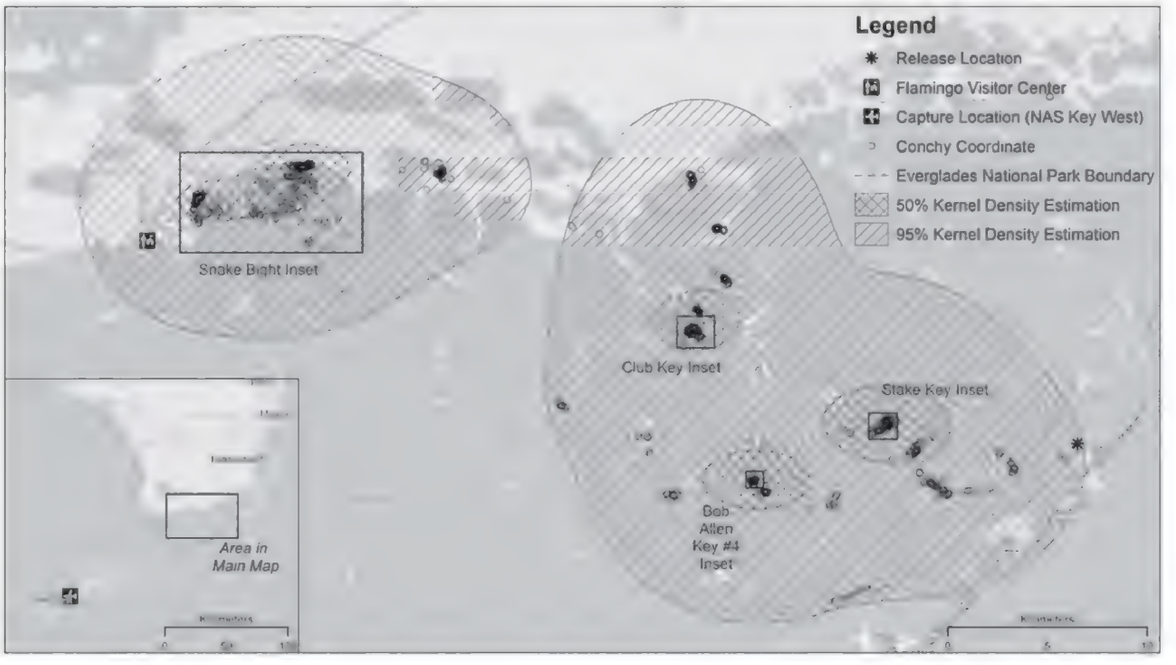

Figure 1. Study area and areas of key importance for Conchy (an American Flamingo) during the 22 months after the second release between Tavernier and Butternut Key, Florida, on 3 December 2015. Most of his satellite fixes were from four main areas: Snake Bight, Club Key, Bob Allen Key #4, and Stake Key. The four boxes in the figure are presented as inset maps in Figure 3.

isolated. After over two weeks of unsuccessful permitted harassment of Conchy to vacate the area by United States Department of Agriculture (USDA) Wildlife Services, we coordinated a capture effort with NAS Key West, Key West Wildlife Center, and USDA Animal Services to prevent potential injury to Conchy, to naval aircraft, and to naval pilots (Gonzalez 2015).

We captured Conchy on 3 October 2015 with a small animal net gun (MagNet Wildlife Capture Services, Flagstaff, AZ, USA). Conchy's generally pink body plumage, pale iris, and gray plumage on the neck and under the wings indicated that he was a sub-adult at the time of capture. Sex determination performed by Avian Biotech International (Tallahassee, FL, USA) identified Conchy as male. We held Conchy in the care of the Key West Wildlife Center (Key West, FL, USA) until initial release. On the day of release, we equipped Conchy with an alphanumeric leg band (blue band with white letters reading US01) and a United States Geological Survey (USGS) metal band (1148-90551) over his left tibiotarsus and a GeoTrack 30-g solar global positioning system platform transmitting terminal (GPS PTT) transmitter (North Star Science and Technology, Oakton, VA, USA; hereafter transmitter) over his right tibiotarsus. We attached transmitter with a modified two-part leg band that we riveted together and padded with a neoprene liner. We released Conchy at Snake Bight, an extensive tidal mudflat in northwestern Florida Bay on the southern coast of the Florida peninsula in Everglades National Park (25.16192°, –80.88204°) at 1724 EST on 20 October 2015.

At ~0800 EST on 27 October 2015, maintenance staff at Everglades National Park found Conchy sitting on a road at the Flamingo Visitor Center (Fig. 1), weak and unable to stand (R. Showler, Everglades National Park, pers. comm.). At that time, we transported Conchy to Zoo Miami's animal hospital for evaluation. Conchy was in poor body condition, was dehydrated, and had a parasitic liver fluke infection. He responded to treatment and regained his strength and body condition over the following five weeks. We housed Conchy at Zoo Miami in isolation and quarantine until his second

release. On 3 December 2015, we released Conchy in Florida Bay near Butternut Key (25.07592°, –80.50526°; Fig. 1), 39.7 km to the southeast of the first release location. Capture, handling, and housing protocols followed applicable regulations and guidelines regarding animal welfare, were approved by Zoo Miami's Institutional Animal Care and Use Committee, and were permitted by Florida Fish and Wildlife Conservation Commission (FWC; LSSC-16-00009A to MD), a federal bird banding permit (23166 to MD), and a United States Fish and Wildlife Service (USFWS) Migratory Bird Treaty Act permit (MB056516) to Zoo Miami.

Telemetry data and analysis.—We programmed the GPS device in Conchy's transmitter to collect four daily GPS locations (at 0000, 0600, 1200, and 1800 EST). Every five days the transmitter turned on for a six-hour cycle to upload GPS data and Doppler determined locations to available satellites within the Argos system. Because of the short duration of the initial release (seven days) and because of concerns about Conchy's health and potential impacts of his health on movement patterns, in this study we excluded data from the first release period and only included data following the second release. We filtered GPS and Doppler data to remove low-quality points. Specifically, we removed one GPS point with an accuracy reading of invalid value, and 302 class 2, 1, A, and B Doppler points with estimated accuracies of >250 m.

To evaluate core areas for Conchy, we used the rhr package in R (Signer and Balkenhol 2015) to produce kernel density estimates and constructed 50% and 95% occupancy polygons. To evaluate habitat characteristics, we categorized areas throughout Florida Bay using a spatial dataset of Florida Bay substrate types produced by the USGS (Prager and Halley 1997). We chose this dataset because nearly all of Conchy's GPS locations were within the domain of this dataset, and because we were interested in quantifying selection for specific substrate types within this area. We used each GPS and Doppler point returned from Conchy's time in Florida Bay as an indication of habitat selection, and compared proportion of time spent by substrate type to the overall representation of that substrate type in Florida Bay. To test habitat selection, we conducted a G-test in the package RVAideMemoire (Herve 2020) in R comparing the hectares of each substrate type in the USGS South Florida Information Access (SOFIA) dataset (availability) to the number of GPS fixes from each substrate type (usage).

To evaluate movement patterns across times of day (diel patterns) and across monthly intervals, we calculated 6-hour step lengths between consecutive fixes. We compiled GPS fixes collected at ~6-hour intervals and excluded sequential GPS points that were more than ~6 hours because intermediate GPS fixes were not obtained. We calculated the distance moved between these sequential GPS locations in ArcGIS Advanced 10.6.1 (Esri, Redlands, CA, USA). To examine differences in step lengths between times of year and times of day, we used a general linear model with time of day and month-year as categorical predictor variables with an interaction term, and with log-transformed 6-hour step length as a response variable.

To evaluate reporting probability via eBird, we queried the eBird community science portal for reports of American Flamingos within Florida Bay and surrounding ecosystems for the period during which Conchy's transmitter returned GPS points within Florida Bay. This reporting probability is conceptually similar to detection probability, with the distinction that detection probability classically assumes that sampling intensity is known. We examined each eBird checklist for any indication that a sighted flamingo was banded or was equipped with a transmitter. For each report that contained photographic records, we examined photographs for evidence of a band or transmitter. We sorted eBird records into three confidence categories (definitive, probable, or possible) that an observed flamingo was Conchy. Definitive reports included photographs where Conchy's band or transmitter were visible and there was no doubt that the flamingo reported was Conchy. Definitive reports represent our most conservative estimate for reporting probability, and can be interpreted as the probability that a single individual flamingo

will be reported and identified over a given time period. Probable reports included both definitive reports and reports that did not include a photograph with a visible band or transmitter but stated that a flamingo was banded or tagged. These reports are very likely to be of Conchy because Conchy was the only known flamingo in Florida Bay with a transmitter or radio during this study period, though this is a less conservative estimation of reporting probability than our definitive category. Possible reports included all sightings of flamingos over the time period, regardless of evidence of band or transmitter (including all definitive and probable reports), and is best interpreted as the probability that any individual of the species will be documented, given that at least one individual is known to be present.

For each checklist, we extracted the date, time, and location. We assigned a value of one to each time point with a report and a value of zero for each time period with no report. We calculated reporting probability for each of our three confidence categories (definitive, probable, and possible) over three temporal scales (daily, monthly, and annually). For our possible reporting probability, for each point estimate of reporting probability, we estimated 95% binomial confidence intervals as a statistical indication of the range of possible values surrounding our point estimates. Finally, from each eBird report we also extracted the number of flamingos because this provides evidence for group size and social structure.

RESULTS

Conchy's transmitter returned GPS and Doppler coordinates between 3 December 2015 and 24 September 2017. Our resulting database included 1,265 GPS locations and 168 Doppler locations over the 661-day period. Of these points, we were able to construct 914 6-hour step lengths. The number of telemetry points returned per month was variable, ranging from 108 points in May 2016 and a low of 4 points in February 2017 (Fig. 2). There was a general low point in number of telemetry points between September 2016 and May 2017.

Habitat selection and movement patterns.—Within 24 hours after release, Conchy flew from the release site near Butternut Key to a mudflat inside Stake Key, which was fringed by mangroves (Figs. 1, 3A, 4A), and most transmitter points returned through December 2015 and January 2016 were from this interior mudflat (Figs. 2, 3A, 4A). In February 2016, Conchy began to visit nearby keys with similar interior mudflats (including Club Key and Bob Allen Key #4; Figs. 1, 2, 3B, 3C, 4A), where he spent considerable time through much of April to September 2016 (Figs. 2, 4B, 4C, 4D). In March 2016, we received the first transmitter coordinate from Snake Bight (Figs. 2, 3D, 4B), although ≤15% of points in any month were returned from Snake Bight until November 2016 (Fig. 2). From November 2016 through May 2017, nearly all of the returned coordinates were from Snake Bight (Figs. 2, 4D, 4E, 4F). We received only a single point from Club Key in June 2017 and continued to receive points from a mudflat within Bob Allen Key #4 in June through September 2017 (Figs. 2, 4G, 4H). The last coordinates returned from Conchy's transmitter were on 24 September 2017 from Snake Bight (Fig. 4H).

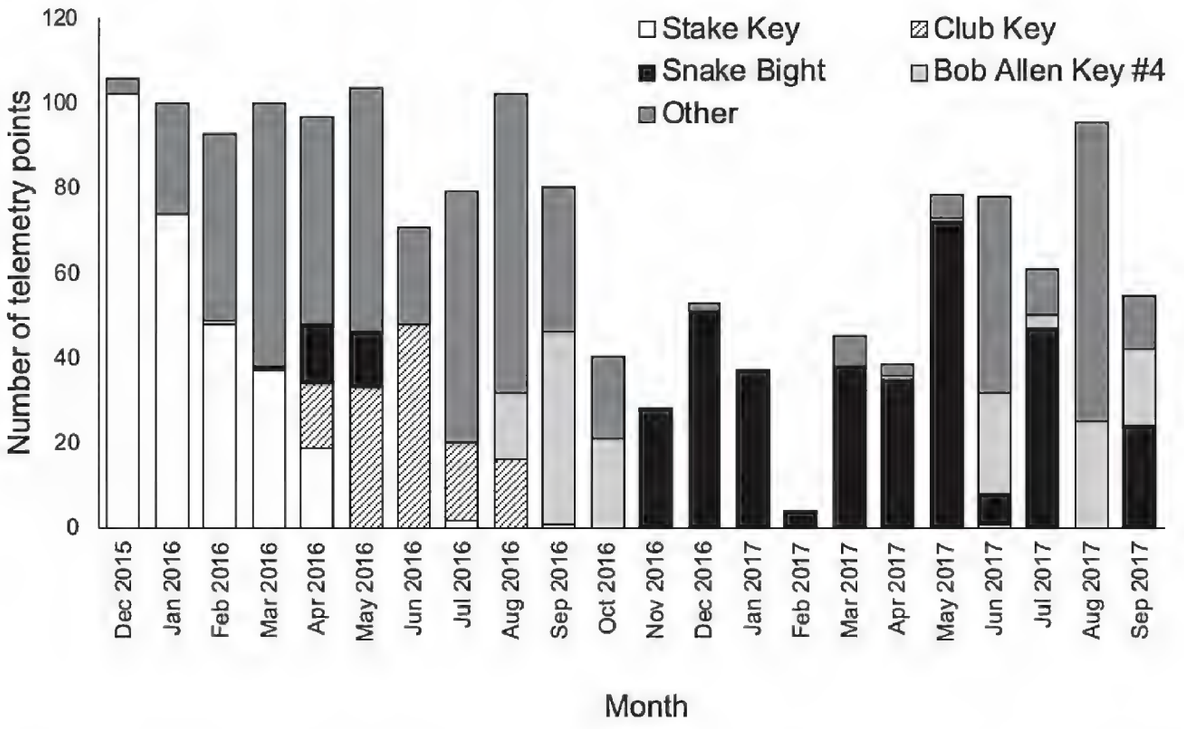

Figure 2. Usage of several key areas used by Conchy (an American Flamingo) in Florida Bay, Florida, illustrated by month. His movement patterns indicate clear temporal shifts in areas used.

Over the entire 22-month period, all but three of the GPS and Doppler fixes were within Florida Bay. One point on 24 May 2016 was in Biscayne Bay near Turkey Point, and two points were just east of Key Largo in January and June 2016. Our 50% core area contour identified four polygons encompassing a total area of 5,704 ha (Fig. 1). One polygon ranged from Snake Bight to Santini Bight, with most points spread between Snake Bight and Garfield Bight (Figs. 1, 3D). The three other polygons each included Stake Key, Club Key, and Bob Allen Key #4 (Figs. 1, 3A, 3B, 3C). The 95% core area contours included two larger but separate polygons, encompassing 66,796 ha (Fig. 1).

There were clear differences in types of substrate used compared to their availability (Figs. 5, 6; G-test, df = 9, G = 1,731, $P < 0.001$). Conchy's transmitter returned far more points on land (including interior mudflats on Florida Bay keys) or on mud bank suite substrates, although these areas only included 29.0% and 11.8% of the area within Florida Bay, respectively. Conchy avoided open water, mixed bottom suite, and hardbottom suite substrates, and avoided all densities of seagrass.

Conchy's 6-hour step lengths varied by month (Figs. 4, 7; $F_{21,871} = 9.52$, $P < 0.001$) and time of day (Fig. 7; $F_{1,871} = 5.94$, $P = 0.015$), but there was no interaction effect between month and time of day ($F_{20,871} = 14.63$, $P = 0.086$). Conchy's travel distances were lowest in December 2015, January 2016, November 2016, December 2016, and

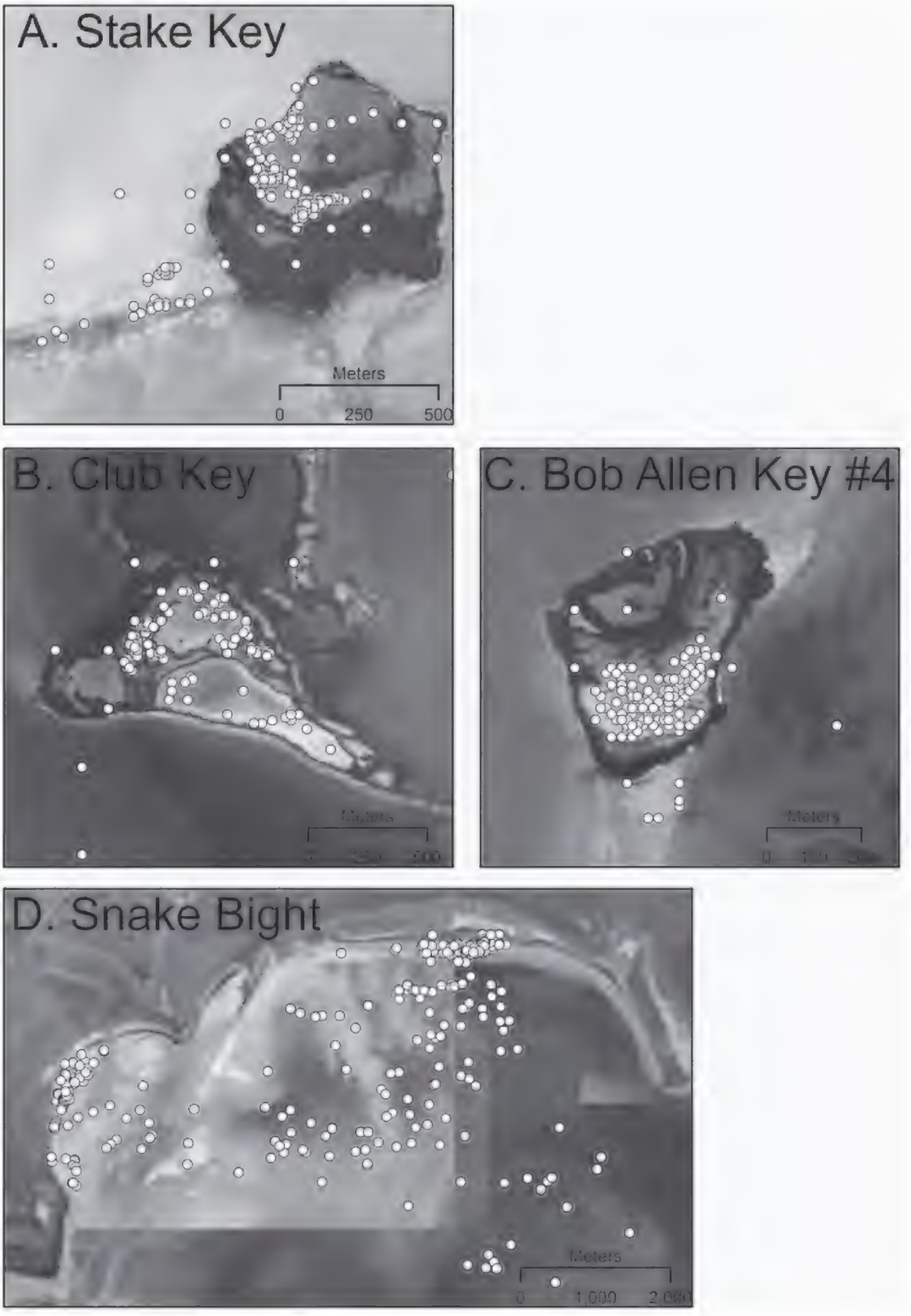

Figure 3. Satellite imagery of four key areas used by Conchy (an American Flamingo) in Florida Bay, Florida, 2015–2017. The three keys shown here (Club Key, Bob Allen Key #4, and Stake Key) each has an interior mudflat and is ringed by a mangrove berm, and Snake Bight is an extensive tidal mudflat in northwestern Florida Bay.

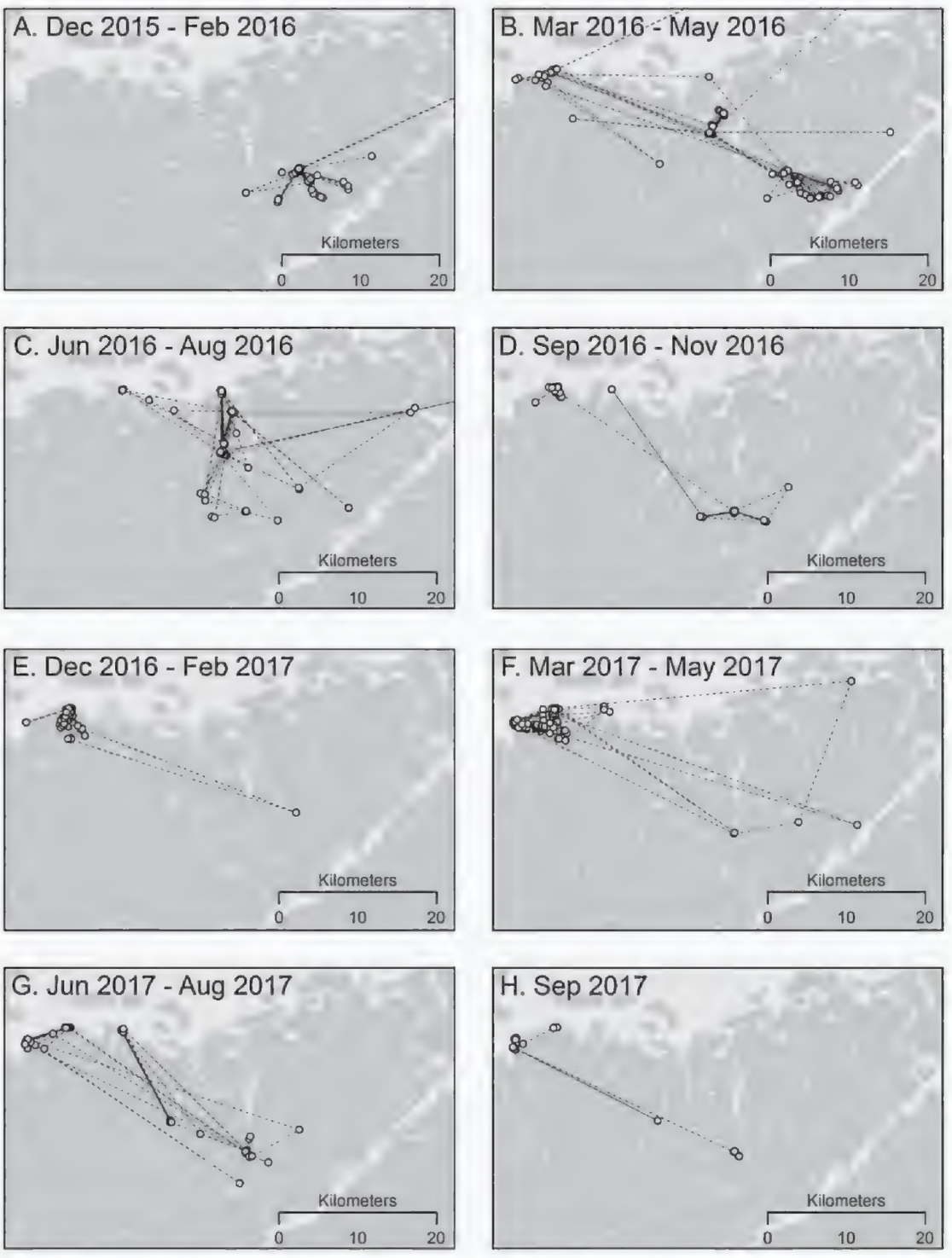

Figure 4. Habitat selection and movement patterns by Conchy (an American Flamingo) in Florida Bay, Florida, as listed by 3-month intervals: A) December 2015–February 2016, B) March 2016–May 2016, C) June 2016–August 2016, D) September 2016–November 2016, E) December 2016–February 2017, F) March 2017–May 2017, G) June 2017–August 2017, and H) September 2017.

September 2017. His greatest average distances were in July and August 2016, July and August 2017, and April 2016. Conchy's greatest travel distances were between 1200 and 1800 hours, and his distances were considerably lower between 0000 and 1200 hours (Fig. 7).

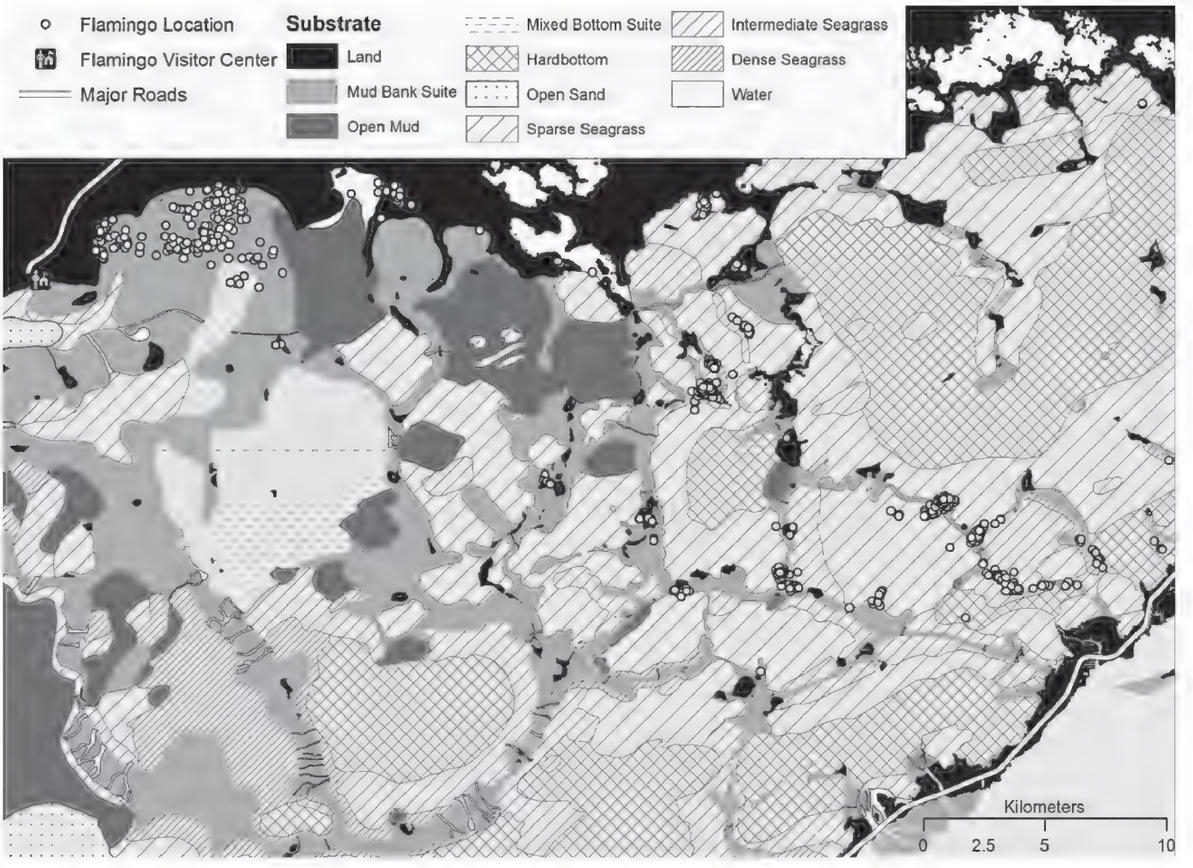

Figure 5. Habitat selection of Conchy (an American Flamingo) in Florida Bay, Florida, 2015–2017, as a function of substrate type. The most important substrate type was mud bank suite followed by land. Conchy selected against remaining substrate types.

Reporting probability through eBird.—Our review of eBird reports yielded 13 definitive Conchy reports, 4 probable Conchy reports, and 9 possible Conchy reports (Table 1). Of the definitive and probable Conchy reports, only two observations were from 2016 and the remainder were from 2017. One definitive report and one probable report were at or near Stake Key (4 January 2016 and 12 April 2016). One possible report was west of Key Largo in December 2016. All 22 remaining reports were from Snake Bight over a relatively short interval between 1 May and 29 July 2017, and 17 reports were from July 2017 alone. During this period of high observations, there were even multiple independent reports from the Snake Bight on the same day (Table 1).

Over the 661 days that Conchy was known to be in Florida Bay, our estimates for daily reporting probabilities were <4% for all confidence categories (Fig. 8), and our 95% confidence intervals indicated that even our highest confidence interval for possible reports was <5%. At the monthly scale, over the 22 months that Conchy was in Florida Bay, monthly reporting probability ranged 13.6% for our most conservative confidence category to 27.3% for our most lax reporting category. For each confidence category, our yearly reporting probability was 100%,

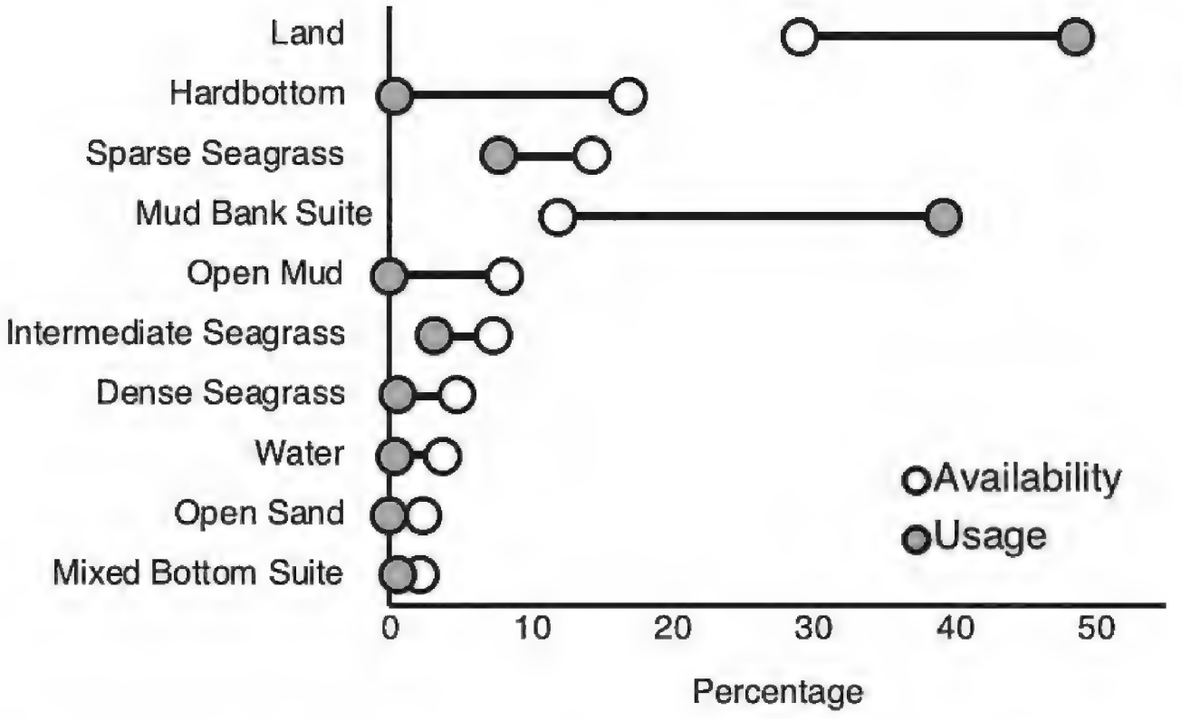

Figure 6. Substrate types selected by Conchy (an American Flamingo) in Florida Bay, Florida, 2015–2017. Conchy strongly preferred land (including impounded ponds on islands) and mud bank suite bottom types.

although our confidence intervals ranged from 15% to 100%, which indicated very low confidence in estimating reporting probability at this temporal scale.

The eBird portal provided additional data on the number of flamingos sighted with Conchy (Table 1). The two observations in 2016 showed him alone, but during May–July 2017 he was typically reported with two additional flamingos.

Discussion

We obtained the first data for habitat selection of an American Flamingo in Florida Bay, and found that flamingos may reside in Florida Bay for nearly two years. We identified four key areas (Snake Bight, Stake Key, Bob Allen Key #4, and Club Key), and two major landscape features (tidal mudflats and interior mudflats) used by this flamingo and others in his small group. We obtained the first satellite telemetry data for this species and identified daily and seasonal differences in movement patterns. Finally, we concluded that eBird has low probability for reporting flamingos, despite their large size and conspicuous nature.

Although our study is restricted to habitat selection and movement patterns of a single individual, Conchy's satellite transmitter returned data that improves our understanding of flamingo habitat selection in Florida Bay. Conchy's transmitter functioned for nearly two

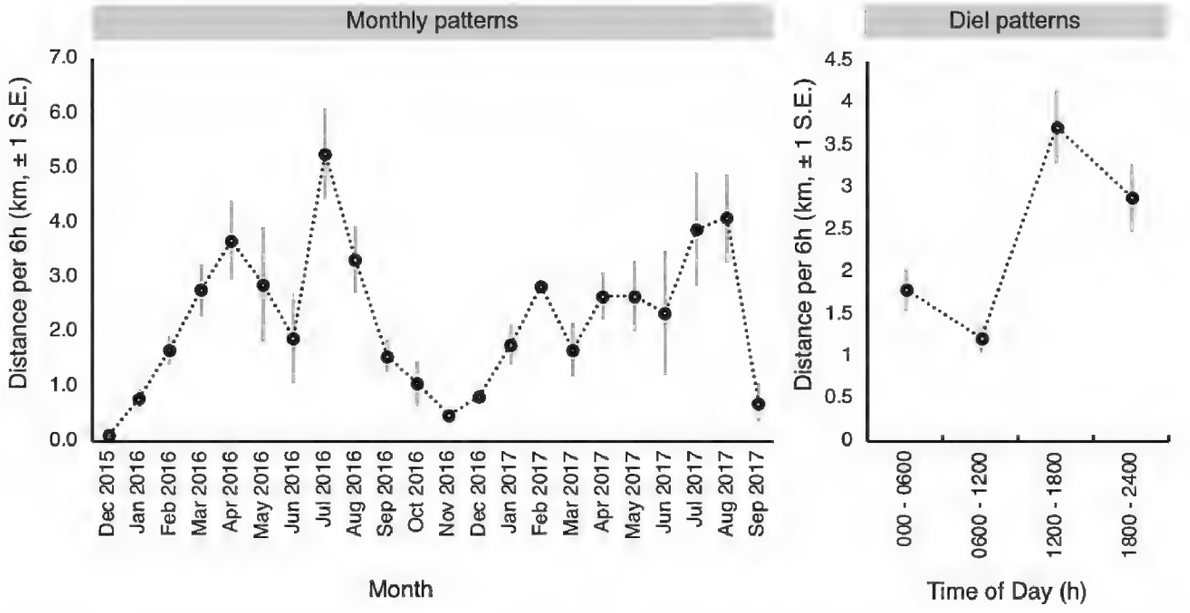

Figure 7. Distances traveled by Conchy (an American Flamingo) in Florida Bay, Florida, 2015–2017, over 6-hour intervals between subsequent location fixes. Conchy moved greater distances in summer months than in winter, and moved greater distances at night than during the day.

years, although the number of GPS points was temporally variable and appeared to decline over the study period (Fig. 2). Likely, the transmitter or antennae suffered some damage from corrosion in the saline environment of Florida Bay and the interior mudflats, or lost the ability to generate solar power through biofouling of the solar panel on the transmitter. However, it is also possible that many GPS fixes could not be obtained if Conchy was foraging in waters that submerged the GPS device or satellite transmitter.

Connectivity and long-distance dispersal.—Ultimately, this study failed to indicate any linkage between Florida's flamingos and foraging or nesting locations outside of Florida. To date, the only evidence for movement of flamingos from the Caribbean to Florida comes from three individual flamingos banded as chicks in the Yucatan of Mexico (Galvez et al. 2016, Welsh and White 2019). However, recent banding programs have been more intensive in the Yucatan than in Cuba or the Bahamas (Arengo and Childress 2004, Galvez et al. 2016), both of which are closer to Florida and have larger overall population sizes. It is unlikely that all flamingos in Florida Bay are of Mexican origin.

Movement between Florida and major nesting areas in the Yucatan, Cuba, and The Bahamas is certainly within normal range of long-distance flights for flamingos (Allen 1956). For example, young flamingos banded in the Yucatan have been sighted in Cuba, Texas, and the Caiman Islands (Galvez et al. 2016). Flamingos banded in Great Inagua in The Bahamas have been resighted in Hispaniola and throughout Cuba (Sprunt 1975). Amat et al. (2005) conducted satellite

Table 1. Sightings of American Flamingos as reported by eBird during the period that Conchy (an American Flamingo) was within Florida Bay (3 Dec 2015–24 Sep 2017). Conchy was sighted on 13 definitive occasions, 3 probable occasions, and 10 possible occasions. A large proportion of sightings were over a 2-month period in June and July 2017.

Date	Observer(s)	Locality	Confidence category	Supporting evidence for confidence category	Number of Flamingos Observed	Citation
4 Jan 2016	P. Frezza and H. Brewer	Stake Key	Definitive[a]	No, cropped from photo[a]	1	https://ebird.org/checklist/S26690793
12 Apr 2016	E. Cantelmo	West of Stake Key	Probable	Description says band or bandage on leg	1	https://ebird.org/checklist/S28902226
19 Dec 2016	T. Davis	Moonbay, Key Largo	Possible	None	1	https://ebird.org/checklist/S33094146
23 Dec 2016	A. Washuta	Snake Bight	Possible	None	2	https://ebird.org/checklist/S33185298
1 May 2017	M. M.	Snake Bight	Probable	Description states band on leg	2	https://ebird.org/checklist/S36466856
22 Jun 2017	B. Showler	Snake Bight	Possible	None	1	https://ebird.org/checklist/S37740114
30 Jun 2017	B. Chou and T. Ford-Hutchinson	Snake Bight	Definitive	Band readable in photograph	3	https://ebird.org/checklist/S37887845
1 Jul 2017	Sam Saunders	Snake Bight	Possible	None	3	https://ebird.org/checklist/S37913859
1 Jul 2017	A. Trentler and B. Trentler	Snake Bight	Definitive	Band and transmitter visible in photograph	3	https://ebird.org/checklist/S37912307
1 Jul 2017	J. Baureister	Snake Bight	Probable	Checklist indicates one flamingo tagged	3	https://ebird.org/checklist/S37919597

[a]In this instance, the observer worked with one of the authors of the present study (JJL) and intentionally cropped the band out of the photograph to prevent undesired attention paid to Conchy by birders in the period shortly after Conchy's release.

Table 1. (Continued) Sightings of American Flamingos as reported by eBird during the period that Conchy (an American Flamingo) was within Florida Bay (3 Dec 2015–24 Sep 2017). Conchy was sighted on 13 definitive occasions, 3 probable occasions, and 10 possible occasions. A large proportion of sightings were over a 2-month period in June and July 2017.

Date	Observer(s)	Locality	Confidence category	Supporting evidence for confidence category	Number of Flamingos Observed	Citation
1 Jul 2017	S. Cotrell	Snake Bight	Definitive	Band code readable in photograph	3	https://ebird.org/checklist/S37903726
4 Jul 2017	S. Olsen	Snake Bight	Definitive	Band visible from photograph	3	https://ebird.org/checklist/S37963315
8 Jul 2017	E. Ahlbrandt	Snake Bight	Possible	None	3	https://ebird.org/checklist/S38036464
8 Jul 2017	C. Hill and M. Hill	Snake Bight	Definitive	Transmitter visible in photograph	3	https://ebird.org/checklist/S38076014
8 Jul 2017	B. Galvin and P. Galvin	Snake Bight	Possible	None	3	https://ebird.org/checklist/S38150128
8 Jul 2017	S. Zuckerman	Snake Bight	Definitive	Band readable in photograph	3	https://ebird.org/checklist/S38035778
12 Jul 2017	T. Witt	Snake Bight	Possible	None	3	https://ebird.org/checklist/S38110342
16 Jul 2017	N. Frade	Snake Bight	Definitive	Band readable in photograph	3	https://ebird.org/checklist/S38170424
21 Jul 2017	A. Yappert	Snake Bight	Definitive	Band readable from photograph	3	https://ebird.org/checklist/S38250405
22 Jul 2017	Y. Morelle	Snake Bight	Possible	None	3	https://ebird.org/checklist/S38264199

[a]In this instance, the observer worked with one of the authors of the present study (JJL) and intentionally cropped the band out of the photograph to prevent undesired attention paid to Conchy by birders in the period shortly after Conchy's release.

Table 1. (Continued) Sightings of American Flamingos as reported by eBird during the period that Conchy (an American Flamingo) was within Florida Bay (3 Dec 2015–24 Sep 2017). Conchy was sighted on 13 definitive occasions, 3 probable occasions, and 10 possible occasions. A large proportion of sightings were over a 2-month period in June and July 2017.

Date	Observer(s)	Locality	Confidence category	Supporting evidence for confidence category	Number of Flamingos Observed	Citation
22 Jul 2017	D. Pettee and D. Bernstein	Snake Bight	Definitive	Transmitter visible in photographs	3	https://ebird.org/checklist/S38268436
23 Jul 2017	D. Olavarria and R. Hattaway	Snake Bight	Definitive	Transmitter visible in photographs	3	https://ebird.org/checklist/S38292048
23 Jul 2017	N. Stuart	Snake Bight	Possible	None	3	https://ebird.org/checklist/S38288652
24 Jul 2017	K. Peart	Snake Bight	Definitive	Transmitter visible in photographs	3	https://ebird.org/checklist/S38307937
27 Jul 2017	E. Rottino, D. Rottino, D. Rottino, T. Rottino	Snake Bight	Definitive	Band readable from photograph	3	https://ebird.org/checklist/S38353310
29 Jul 2017	M. Zimmerman	Snake Bight	Possible	None	2	https://ebird.org/checklist/S38451588

[a]In this instance, the observer worked with one of the authors of the present study (JJL) and intentionally cropped the band out of the photograph to prevent undesired attention paid to Conchy by birders in the period shortly after Conchy's release.

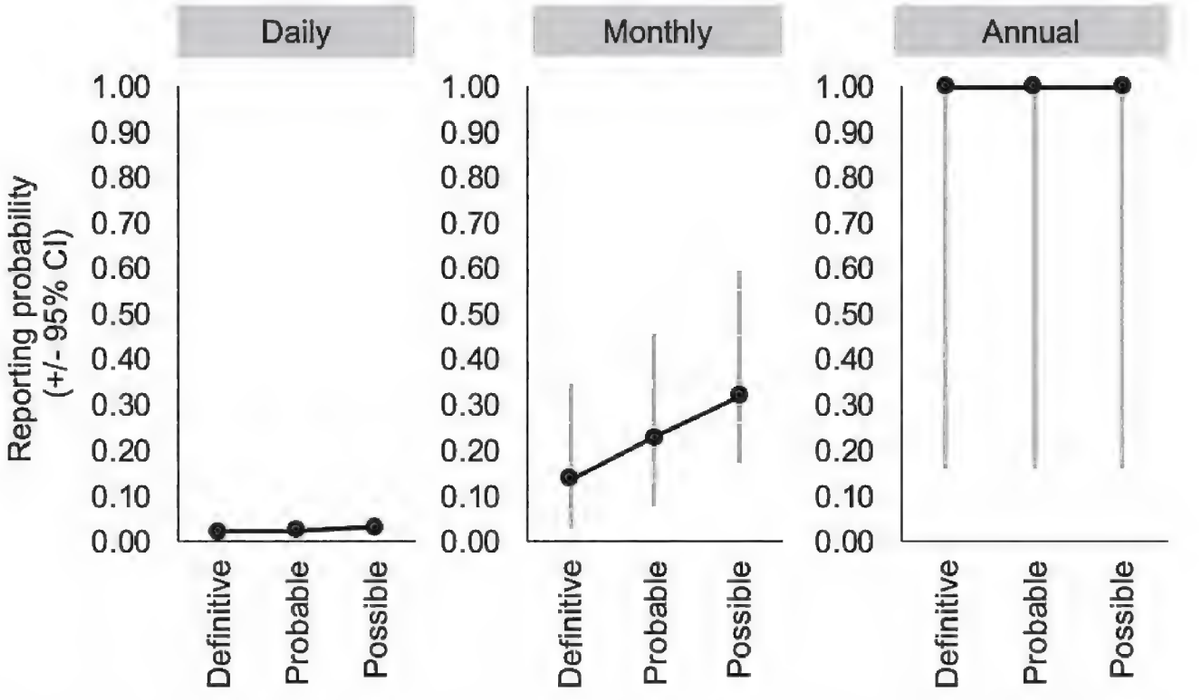

Figure 8. Reporting estimates for Conchy (an American Flamingo) in Florida Bay, Florida, 2015–2017, from the community science portal eBird at three temporal scales (daily, monthly, and annual) and three reporting confidence intervals (definitive, probable, and possible). Each value is the percentage chance of Conchy being reported in the reporting time interval. Error bars for the daily reporting probabilities are obscured by marker points because confidence intervals for these metrics were very small.

telemetry with Greater Flamingos (*Phoenicopterus roseus*) and found that nesting adults foraged up to 200 km from the nesting colony, that flamingos may travel up to 400 km in a single night, and that adults wintered up to ~2,000 km away from nesting sites. Such long travel distances would place Florida within foraging range of nesting colonies in northern Cuba, and within wintering range of all nesting populations of American Flamingos except for the Galapagos Islands.

Habitat associations.—Our analysis using bottom types should provide a preliminary indication of habitat associations. Conchy largely used substrates identified as mud bank suite, which includes areas dominated by carbonate mud or sand where water depth is generally less than 0.6 m in depth. The largest contiguous spatial unit of this substrate in Florida Bay is at Snake Bight, with smaller patches of mostly submerged mud banks spanning between keys and forming divisions between basins in the bay (Fig. 5). The land category in our analysis included mangrove berms ringing Florida Bay Keys and the interior mudflats themselves. These substrate classifications are certainly simplifications that obscure important habitat features for flamingos, yet we are unaware of other existing classification layers for Florida Bay, and reclassification *de novo* is beyond the scope of this study.

Still, Snake Bight and the interior mudflats and salt ponds inside Florida Bay keys differ in a number of important ways. Snake Bight is an expansive (39 km^2) carbonate mud flat that experiences tidal exchange with Gulf of Mexico and has an average daily tide of 1 m that alternately exposes the entirety of the Bight's mud bottom, followed by flooding of up to 0.6 m. Snake Bight is typical in this respect to other estuaries (e.g., Celustun Lagoon in the western Yucatan Peninsula of Mexico) where large concentration of flamingos can be found roosting on large mud banks at low tide (J. J. Lorenz, Audubon Florida, pers. obs.). Conchy's frequent use of tidal mudflats at Snake Bight confirms historical observations of this area as important habitat for flamingos, going back to the mid-nineteenth century (Wurdemann 1860, Scott 1890, Ingraham 1893, Allen 1956).

The interior mudflats are ecologically rather distinct from Snake Bight. Their hydrology is functionally independent from Snake Bight, and from each other. Eastward across Florida Bay, a series of interlocking mud banks and island chains dampens tidal effects, resulting in a diurnal tidal signature <1 cm in northeastern Florida Bay, which is easily masked by even mild winds (Holmquist et al. 1989, Wang et al. 1994). The three islands that Conchy used frequently were further protected from tidal influence by a mangrove fringe that is found on a berm along the perimeter of the islands (Wingard et al. 2019; Fig. 3). The interior mudflats of these islands lay slightly below sea level (Wingard et al. 2019) and are generally isolated from direct surface hydrologic communication with the surrounding Bay. The mudflats dry through evapotranspiration and periodically reflood with Bay water because of strong wind tides (J. J. Lorenz, pers. obs.) or rainfall (G. L. Wingard, USGS, pers. comm.). Although these interior mudflats are poorly studied, salinity normally ranges from 0 to 75 psu (G. L. Wingard, unpublished data), and the mudflats can fluctuate from being essentially dry with isolated and extremely hypersaline puddles (165 psu) to fully flooded with fresh water or ambient salinity bay water in a 24-hour cycle (G. L. Wingard, unpublished data; J. J. Lorenz, pers. obs.). In the event that reflooding events become infrequent, the mudflats gradually dry out over days or weeks depending on the volume of water. Although these biologically harsh conditions keep the interior ponds free of vascular plants, algal communities often form mats that cover the bottom (Wingard et al. 2019). Many of the small fishes and crustaceans typically found throughout Florida Bay's mangrove forests are well adapted to these rapidly fluctuating conditions (Lorenz and Serafy 2006) and are frequently observed within the ponds (J. J. Lorenz, pers. obs). This prey availability probably makes them suitable for foraging by flamingos, but the foraging conditions would be very different than those found at Snake Bight. Use of these interior

mudflats as foraging habitat by flamingos has not been previously described.

Diel and seasonal movement patterns.—Conchy's movements varied seasonally and by time of day. The shortest step lengths were in winter months (Figs. 4, 7), and we found considerably longer average distances traveled during summers. We also found that Conchy's movement distances varied by the time of day with much longer distances in the afternoon and evening than before noon. Presumably, these movements indicate tracking of prey availability over time, and are likely influenced by variable hydroperiods of the key areas identified above.

Reporting probability.—Our review of eBird records indicated that reporting probability was low, and was highly inconsistent over space and time. Thirteen of seventeen definitive or probable sightings over a ~2-year period were within a single month at Snake Bight. This was likely the consequence of an initial eBird report in June 2017 that motivated numerous observers to seek flamingos from a known and relatively accessible location. Although a large amount of Conchy's time was spent in interior mudflats, there was only one definitive observation over the 22-month period in these areas. All but three keys in Florida Bay are closed to visitation by people according to Everglades National Park regulations. Clearly, community science observations alone are likely to dramatically underestimate the number of flamingos within Florida Bay and may not produce accurate information on habitat associations. The very low reporting probability we estimated is somewhat surprising given flamingos' large body size, conspicuous coloration, and gregarious nature.

The biases we identified in community science reporting probabilities have implications for our understanding of the number and distribution of flamingos in Florida. Although most flamingos are likely to go unreported, flamingos that are observed may be substantially overcounted if they reside in a small area for an extended period. Unfortunately, there are no ongoing monitoring efforts for flamingos in Florida, despite considerable attention that is devoted to monitoring other species of wading birds in South Florida by state and federal agencies (Frohring et al. 1988, Hunter et al. 2006, Cook and Baranski 2019). We suggest that the low reporting probability we describe underscores a need for systematic monitoring, and we suggest that the habitat associations we identified herein could help optimize a monitoring program in Florida Bay.

Flamingos as year-round residents in Florida.—We did not expect at the onset of our study that Conchy would reside within Florida for nearly two years. Rather, we assumed that flamingos are short-term stopovers in Florida Bay, or intermittent foragers that would

return within weeks to foraging areas outside of Florida. Still, Conchy remained in Florida Bay despite two periodic cold snaps (late January 2016 and late January 2017) when air temperatures fell below 8°C. These cold snaps are likely beyond the temperature extremes experienced by flamingos in The Bahamas or the Caribbean, yet they still did not drive Conchy to leave to warmer areas.

There are two likely explanations for Conchy's long residence in Florida Bay. First, it is possible that his residence is atypical and uninformative because as a subadult bird upon capture he may have relied on older and more experienced birds to lead him in long-distance movements. Still, after only his second month post-release, he was consistently seen with adult flamingos that may have provided guidance on flyways or regional movement patterns. Alternatively, it is possible that Florida Bay is still habitat for flamingos—at least of sufficient quality for foraging by small groups. Clearly, historical evidence suggests that Florida Bay did once serve as foraging (and likely nesting) habitat for large flocks of flamingos (Wurdemann 1860, Scott 1890, Holt 1924, Allen 1956, Whitfield et al. 2018), yet human alteration of South Florida's hydrology has had major impacts on the function of Florida Bay (McIvor et al. 1994, Fourqurean and Robblee 1999, Lorenz 2014, Wingard et al. 2019), which may influence contemporary foraging and nesting opportunities. It is currently not possible to empirically evaluate quality of habitat in Florida Bay to foraging or nesting for groups of flamingos, although this seems to be a necessary step to facilitate population recovery.

Other rare flamingo information may help put Conchy's long residency in context. One flamingo (band DFJV) sighted in Everglades National Park in October 2002 was banded in the Yucatan and was sighted again in the Yucatan between 2003 and 2006, which suggests a short-term foraging visit to Florida from Mexico. A slowly growing group of flamingos (up to 147 individuals in 2014) has been observed in Stormwater Treatment Area 2 (Palm Beach County, Florida) during the spring for several decades but has no observations reported for other seasons. Likely, these are periodic foragers visiting from outside of Florida, despite the low and spatially variable reporting probabilities we estimate in this study. Finally, an isolated flamingo in the Florida panhandle has been sighted consistently between 1 November 2018 and the date of this writing (December 2020). It seems reasonable to assume that this is the same flamingo, and its residency is at 25 months as of December 2020—exceeding the duration of Conchy in Florida Bay. Collectively, these data suggest that coastal environments throughout Florida appear climatically suitable for flamingos and that foraging habitat is available. Other species of flamingos do not appear exclusively restricted to tropical climates. Greater Flamingos will

winter in southern Europe and multiple Andean species of flamingos live in high elevations where nightly temperatures drop below freezing (Caziani et al. 2007). It is unclear if Florida's coastal environments are outside of the thermal tolerances for American Flamingos—and Conchy's persistence through two cold snaps seems to confirm this point.

Regardless of why Conchy would remain in Florida Bay for nearly two years, increasing reports of flamingos in Florida and documentation of year-round residency are concordant with areas outside of Florida where flamingos appear to be recolonizing their historical foraging and nesting habitat decades after populations were decimated by hunting. For example, flamingos were extirpated from their historical nesting areas in Hispaniola but have been present for decades as year-round residents despite repeated failed attempts at nesting (Wiley 1979, Paulino et al. 2011). Flamingos were also extirpated from Isla Margarita in Venezuela, yet have recolonized and resumed nesting after decades of absence (Sanz D'Angelo 2020). To our knowledge, there is no evidence for nesting of flamingos in Florida since at least the early 1900s (Sprunt 1937, Whitfield et al. 2018), yet the absence of formal monitoring programs and low reporting probability we estimate would make detection of nesting areas unlikely. As urged by Hunter et al. (2006), we argue that systematic monitoring for nesting activity should be a conservation priority for flamingos in Florida.

Transmitter failure and Conchy's fate and future.—Conchy's last transmitter communication was on 24 September 2017. On 10 September 2017, just two weeks before, Hurricane Irma made landfall in the Florida Keys as a powerful Category 4 storm, and its path traversed Florida Bay. High winds and storm surge from Hurricane Irma likely had dramatic impacts on areas Conchy used before the storm. Wingard et al. (2019) showed that Hurricane Irma deposited mud and plant detritus within the interior mudflats, and that storm surge caused a loss of elevation on the surrounding mangrove berms. Snake Bight endured major storm surge as well, which dramatically affected the ecology of that area. Conchy's transmitter showed movement between 11 September and 24 September from Bob Allen Key #4 to Snake Bight and large movements within Snake Bight indicating that he very likely survived Hurricane Irma (Fig. 4H). The disturbances to the Florida Bay ecosystem driven by Hurricane Irma could have motivated a long-distance movement for Conchy (and any other flamingos in his group). It is also possible, though less likely, that Hurricane Irma shifted foraging locations for Conchy and that he has moved into an area of South Florida where reporting probability is even lower than Florida Bay.

After our last transmitter reading, the only circumstantial evidence of Conchy's location at the time of writing this manuscript was an eBird checklist on 5 November 2017: a group of five flamingos reported and photographed at Snake Bight (Forsyth 2017). The photographs are from a considerable distance, and the flamingos in the images are obscured by heat shimmer. Yet inspection of the photographs indicates blue and apparently reflective spots on the leg of one of the flamingos, and we have no explanation for these color patterns other than the blue band and reflective transmitter on Conchy's legs. If our interpretation of this photograph is correct, then Conchy survived after transmitter failure. Further, if Conchy was last sighted with a group of four other flamingos, it increases the probability that he dispersed from Florida with these flamingos directing the path to other foraging or nesting areas.

Though satellite telemetry did not produce information on Conchy's origins, it is possible future observations or analysis may yet provide answers. Flamingos are long-lived and may reach 50 years of age in captivity (Lyngle-Cowland and Lynch 2017) and 49 in the wild (USGS Bird Banding Lab 2020) and have high adult survival. It is plausible that Conchy may be resighted at another location or may reappear in Florida Bay at some point in the future. In Greater Flamingos, juveniles that leave their natal colony for distant foraging grounds in their first or second winter typically return to these same foraging grounds outside of the nesting season later in life (Green et al. 1989). Still, more than three years after his last transmitter coordinates, we have no definitive sightings of Conchy from Florida Bay or elsewhere.

The information we provide herein should help inform evidence-based conservation planning for American Flamingos in Florida, which have surprisingly received virtually no research, conservation, or management attention in the era of modern wildlife conservation. American Flamingos were considered native to Florida in the nineteenth and early twentieth centuries, and remain protected as native species under the United States Migratory Bird Treaty Act of 1918. Their rapid decline in the late nineteenth century generated calls for their protection (Scott 1890, Brodhead 1910). But because the provenance of flamingos in Florida has been uncertain for nearly a century (Allen 1954, 1956; Whitfield et al. 2018), there has been considerable confusion on appropriate management or conservation action for flamingos, and there has been no conservation or management attention by state or federal wildlife agencies. Though a report by USFWS argued that flamingos are a species of regional concern in need of critical recovery attention (Hunter et al. 2006), flamingos have never been considered for protection under the United States Endangered Species Act. At the state level, flamingos are not included in Florida's imperiled species

management plan (FWC 2016), and are entirely absent from Florida's state wildlife management plan, which includes all species native to Florida (FWC 2012). However, as of December 2020, FWC is conducting a biological status review to determine the flamingos eligibility for inclusion under Florida's threatened species laws. We expect that findings from this study—multi-year residency of flamingos in Florida Bay, identification of Snake Bight and interior mudflats as potential key habitat associations, and low reporting probability via community science programs—should be directly relevant to the formation of appropriate conservation and management actions for the species.

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FEATHER MITES FROM TWO PASSERINE BIRDS ON VACA KEY, FLORIDA

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Feather mites are the most abundant and diverse components of avian ectosymbionts and they occur on almost all species of birds (Doña et al. 2019). This brief note reports feather mites taken from two birds found dead under previously described circumstances (Hribar 2019). I collected mites and mounted them on microscope slides with the same methods used earlier (Hribar and Miller 2011). I deposited all mites into the acarology collection of the Zoological Institute, Russian Academy of Sciences, St. Petersburg.

A male Painted Bunting (*Passerina ciris*) was found in Marathon, Florida (Vaca Key) on 3 December 2018. I recovered 61 mites; all were *Trouessartia ciris* (Trouessartiidae). This mite species has only recently been described from Painted Buntings in Georgia (Mironov and Chandler 2020); this is the first report of this mite species from Florida. The true geographic distribution of *T. ciris* is unknown, but it is potentially much broader than Georgia and Florida. For example, Montiel-Parra et al. (2007) reported finding an unidentified *Trouessartia* sp. mite on a Painted Bunting in Mexico. It is nearly certain that the distribution of the mite is the same as that of the host. There are two breeding populations of Painted Buntings, one that occurs in Georgia, Florida, North Carolina, and South Carolina, and another that is found in Arkansas, Kansas, Louisiana, Missouri, Nebraska, Oklahoma, and Texas. Specimens are also known from Alabama, Arizona, New Mexico, and Mississippi. Overwintering populations are found in the Caribbean, Mexico, and Central America as far south as Panama (Storer 1951, Thompson 1991, Yirka et al. 2018).

Also in Marathon (Vaca Key), on 6 May 2019, I removed 29 mites from a Common Grackle (*Quiscalus quiscula*). Dr. Sergei Mironov identified two species, an undescribed *Trouessartia* sp. and *Mesalgoides* sp. near *johnstoni* Spory (Psoroptoididae). Previously, I collected three individual mites from a grackle on Vaca Key. Dr. Mironov identified the mites as an undescribed *Trouessartia* sp., 1 male and 1 female; and one oribatid mite (Oribatei; Hribar 2019). *Mesalgoides* mites can be found in the feathers but also on the skin of the wings; they are more difficult to locate and collect than mites that dwell on the vanes of flight feathers, and as such the genus is poorly known (Mironov et al. 2018). In Florida, mites in this genus have been found on the Brown-headed Nuthatch (*Sitta pusilla*) in Collier County, the Florida Grasshopper Sparrow (*Ammodramus savannarum floridanus*) in Brevard and Osceola counties, and the Pileated Woodpecker (*Dryocopus pileatus*) in Alachua County (Forrester and Spalding 2003).

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FIRST RECORD OF LEAST BITTERN (*Ixobrychus exilis*) ROOSTING IN PINE FOREST CANOPY

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The Least Bittern (*Ixobrychus exilis*) is a secretive marsh bird that is rare to locally common throughout peninsular Florida (FWC 2003, Greenlaw et al. 2014) during the breeding season (mid-March–early August; Stevenson and Anderson 1994). Migrants occur in Florida outside the breeding season, but little is known about the species' migratory behavior and their migratory routes (Poole et al. 2009). Few specimens have been collected from collisions with communications towers, with most of those occurring during spring migration (Crawford 1974, Taylor and Kershner 1986, Stevenson and Anderson 1994). The Least Bittern spends most of its time in dense vegetation in marshes with standing water (Weller 1961, Poole et al. 2009). I report observations of a Least Bittern found during the non-breeding season roosting in the canopy of a pine forest and discuss a possible explanation for this behavior.

On 15 October 2020, I was walking through a ~30-ha slash pine (*Pinus elliottii*) plantation that was surrounded by mixed hardwood hammocks in Newnans Lake State Forest, Alachua County, Florida, when I noticed several songbirds engaging in mobbing behavior. At 1006 h, I heard several Pine Warblers (*Setophaga pinus*) and Carolina Chickadees (*Poecile carolinensis*) calling and converging in the crown of a tall pine roughly 15 m from the trail. I started identifying and counting other birds as they joined the group. At 1007 h, I located a Least Bittern near the outer edge of the tree's crown, perched on a twig near the end of a long branch, just medial from a whorl of pine needles. The bittern perched motionless, in a frozen stance, with its neck and head in an upright posture with bill pointing <10° from vertical. Its outstretched length was comparable to the length of the pine needles, affording it a measure of camouflage. The bird lacked the black crown and black back of an adult male, but I was unable to determine whether the bird was an adult female or a juvenile.

At 1008 h, the flock mobbing the bittern included five Pine Warblers, two Carolina Chickadees, three Tufted Titmice (*Baeolophus bicolor*), one Brown-headed Nuthatch (*Sitta pusilla*), and one Yellow-throated Warbler (*Setophaga dominica*). For the next 5 min, the chickadees scolded loudly and constantly, the other species scolded intermittently, and some individuals moved to within 0.5 m of the bittern. At one point, a male Pine Warbler approached from behind and lightly struck the bittern's rump twice. In response, the bittern contracted its body slightly but otherwise remained completely still in a frozen stance.

By 1013 h, mobbing calls decreased in frequency and intensity and the participants began to leave. I remained at the tree until approximately 1120 h, but the bittern never moved. When two American Crows (*Corvus brachyrhynchus*) flew overhead calling at 1046 h, the bittern briefly puffed up and raised its crown feathers but otherwise remained motionless.

I photographed and marked the tree's location and returned several days later to collect measurements. The roost tree was 30.5 cm in diameter at breast height and approximately 23 m tall. The Least Bittern had perched at a height of about 21.5 m.

I discovered the Least Bittern roughly 800 m from a wetland on Newnans Lake and in the center of a large pine stand. The observation occurred during a time of year when the species migrates south through the northern peninsula of Florida (Greenlaw et al. 2014). Thus, the behavior and location of this bittern were consistent with that of a nocturnal migrant that had touched down during the day for rest. These observations may afford a novel window into the migratory habits of the species, given the paucity of information available about migration for most of the world's 16 species of bitterns. Migration in the Little Bittern (*Ixobrychus minutus*) can be nocturnal or diurnal (Martínez-Vilalta et al. 2020a). Migration is believed to be nocturnal for the American Bittern (Lowther et al. 2009), Great Bittern (*Botaurus stellaris*; Martinez-Vilalta et al. 2020b), and Yellow Bittern (*Ixobrychus sinensis*; Martinez-Vilalta et al. 2020c), and towerkill data in Florida appear to confirm nocturnal migration for the Least Bittern (Crawford 1974, Taylor and Kershner 1986, Stevenson and Anderson 1994).

The Dwarf Bittern (*Ixobrychus sturmii*) has been observed perching in low trees in Africa (Martinez-Vilalta et al. 2020d), but Poole et al. (2009) made no mention of Least Bitterns roosting in trees or shrubs or using woodlands. More information is needed to determine whether Least Bitterns use forested areas regularly as roost sites during migration or whether my observations here reflect atypical behavior of an individual succumbing to the stress of migration.

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ABSTRACTS OF STUDENT PRESENTATIONS AT THE FALL 2020 VIRTUAL MEETING OF THE FLORIDA ORNITHOLOGICAL SOCIETY

12 December 2020
Virtual Meeting via Zoom

The fall 2020 meeting of the Florida Ornithological Society featured our first completely virtual endeavor to ensure the safety of all participants and organizers during the COVID-19 pandemic. Despite the novel medium, students and professionals from around the state and beyond provided informative presentations on current avian research and conservation efforts. Many of the presentations will likely lead to full-length articles, but Abstracts submitted by participants are provided here for members of the Florida Ornithological Society who could not attend the meeting. This also serves as a record of the extent of avian research being conducted state-wide for future scientists and educators.

The Abstracts are listed alphabetically based on the last name of the presenter (who is also listed as the email contact). All author's affiliated institutions are in Florida unless otherwise noted. Student abstracts are listed first and professional abstracts at the end.

Long-term breeding bird community changes in primary forests in the Great Lakes Region of North America

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A recent, high-profile study documenting over three billion birds lost in North America in the past fifty years highlighted the extinction crisis facing our continent's avifauna. Songbirds breeding in the boreal forest represent among the fastest-declining groups. In particular,

boreal bird species associated with cooler climates show ongoing and predicted northward range shifts with climate change, especially as warmer temperatures allow for more southerly breeding species to expand their ranges northward. Such range shifts and associated local extinctions of species are predicted to be most detectable at the southern range limits of boreal species, such as in Michigan's Upper Peninsula. Our study evaluated the temporal changes in the abundance and composition of the breeding bird community at the Huron Mountain Club (Marquette County, MI) by replicating census transects sampled in the 1990s. The Huron Mountain Club (HMC) is a privately owned, relatively untouched, old growth boreal forest that provides an opportunity to isolate the effects of climate change from other factors, such as fragmentation and habitat disturbance. Our preliminary results suggest that species associated with northern breeding habitats, like the Magnolia warbler (*Setophaga magnolia*), are declining at the HMC. There also seems to be stability or emergence of species that breed in both boreal forests and hardwood forests. For birds that have large yearly fluctuations in abundance, such as winter finches, it is difficult to determine species' responses and requires more years of data. Future research in this region should focus on the mechanisms of species turnover and identify important habitat for boreal bird conservation as ranges continue to shift.

The effects of habitat management on Southeastern American Kestrel provisioning rate

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The Southeastern American Kestrel (*Falco sparverius paulus*) is a state-threatened subspecies in Florida and has suffered an 82% population decline in the previous century due in part to fire suppression and loss of nest sites. One of the largest populations of Southeastern American Kestrel resides in a scrub landscape in Ocala National Forest (ONF), which is intensively managed for pulpwood with treatments including prescribed burns and clearcuts. We conducted provisioning rate surveys at kestrel nests and used generalized linear models to assess the relationship between environmental covariates and provisioning rate (minutes between each feeding, selecting for the median value) and the total number of feedings. Provisioning rate and total number of feedings varied among sites for the Southeastern American Kestrel. The best ranked model for total number of feedings had a positive relationship with snag density and a negative relationship with pine cover. The best model for provisioning rate showed a negative relationship with snag density, indicating that as snag density increased, the median time it took to have a successful feeding incident decreased. Concurrent research in ONF has found that snag density is also related to occupancy and daily survival rate of Southeastern American Kestrels. Our research suggests that snag management may also be related to foraging success and provisioning rate. These considerations are important for management decisions concerning scrub landscapes, and similar research is needed in other Southeastern American Kestrel habitats, such as areas where nest boxes are used.

What's up your gut? An experimental test of the microbiota-gut-HPA axis in a wild songbird

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All animals host a gut microbiome, the natural bacterial community in the gastrointestinal tract. Emerging research in mammals shows the gut microbiome is related to host health and important facets of health such as cognitive function, behavior, and the stress response. These relationships are collectively known as the gut-brain-HPA axis. Despite a wealth of research detailing these relationships in rodents and humans, relatively very little is known about this system in birds and other taxa. Building on preliminary results from my dissertation where I found evidence supporting a microbiota-gut-brain axis in a captive Zebra Finch (*Taeniopygia guttata*) population, here I present a summary of my progress in the remainder of my dissertation where I am studying the microbiomes of wild Northern Cardinals (*Cardinalis cardinalis*) in a South Florida population. I will experimentally provoke a chronic stress response in wild Northern Cardinals to test for effects of stress on the gut microbiome, body condition, beak coloration, and vocal performance. Better understanding a wild bird's gut microbiome, and how that community relates to stress, health, and behavior, is a first step to using microbiomes as a tool to index wild animal population health, and ameliorate stress and improve the health of animals in captive breeding and rehabilitation and release programs.

MOVEMENT AND BEHAVIOR OF AMERICAN WHITE IBIS IN SOUTH FLORIDA

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American White Ibis are a common sight in urban and suburban areas in South Florida but also live in wetlands and other natural areas. Urban areas have the potential to change ibis behavior because ibis in urban areas are in different land cover types, like lawns, and interact with people. We will discuss two studies of ibis movement and behavior in and near public parks in Palm Beach County, Florida. First, we will ask how ibis move among urban and natural sites in the landscape, and how they contribute to connections between these two site types. Next, we explore how anthropogenic food provisioning influences ibis behavior in urban areas. Together these results show that ibis behavior and movement differ at both landscape and local scales. We conclude that these patterns can help understand how urbanization affects wading birds.

ABSTRACTS OF NON-STUDENT PRESENTATIONS

New home range methodology for the Red-cockaded Woodpecker

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Home range estimates are commonly used to inform management decisions and analyze the habits of animals. The Red-cockaded Woodpecker (*Dryobates borealis*; RCW) is an endangered species that forms cavities in close proximity to each other in living pine (hereafter cluster) for daily roosting. Historically, researchers resight uniquely color-banded birds from dawn to dusk on a daily basis (termed all-day-follows) to collect locations for home range analysis. Although effective, this method is time-consuming and can be very labor intensive to derive

annual home ranges in a manner that is homogenous through time. In this study, I present an alternative method of collecting RCW locations using randomly generated survey locations. When an RCW is detected at a survey location, the researcher follows the RCW group and records locations. Thus, a researcher can record unbiased locations of RCWs without following a single group for an entire day. To test this method, I sampled 22 clusters across three properties in the Red Hills Region of South Georgia and North Florida. For each survey, I generated ten random locations within pine forests spaced 200 m away from each other in an 800-m radius from the cluster center. Preliminary results from 2019 annual home ranges offer comparable estimates to those of all-day follows in previous studies in this region (32.5 ± 3.92 ha [95% kernel density estimation ± SE]) with RCWs detected during 51–100% of surveys. This method can help improve efficiency of RCW home range studies and provide a useful tool for further research.

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VIABILITY OF A FLORIDA SCRUB-JAY (*Aphelocoma coerulescens*) POPULATION IN NORTH-CENTRAL FLORIDA

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Abstract.—Exceedingly limited breeding range, dispersal ability, and habitat connectivity tend to characterize species as at risk of extinction. A species endemic to Florida, the Florida Scrub-jay (*Aphelocoma coerulescens*; scrub-jay), exemplifies these characteristics, which have contributed to its alarming, roughly 90% population decline since the 1800s. To help stop or reverse this ongoing decline, land managers should communicate scrub-jay population trends to facilitate conversations about the effectiveness of respective management regimes within and between metapopulations and to identify regions in particular need of conservation focus. We report scrub-jay population survey data at Seminole State Forest (SSF) in north-central Florida between 2008 and 2016. Our data suggest a population high in 2008 (n = 137), a low in 2015 (n = 90), and 97 individuals in 2016. Using these data, we modeled a simple population viability analysis which can be easily replicated, and interpreted with caution, by those with similar count data. Our analysis indicated a 0.78 extirpation probability of scrub-jays at SSF within 100 years. Relatively small, simulated increases in population growth substantially decreased this probability. Given that SSF scrub-jays likely constitute the largest portion of their metapopulation, continued and perhaps increased support for management efforts at SSF may be required for metapopulation persistence. Our results highlight the importance of population monitoring with respect to validating current, and identifying the need for future, management efforts.

Key words: avian, corvid, population trends, Seminole State Forest, wildlife management

For species that are highly habitat-specialized, habitat loss is currently the most common ultimate cause of extinction (Fahrig 1997, Owens and Bennett 2000). Proximate causes for extinction other

than habitat loss include stochastic (random) processes regarding demography (e.g., births and deaths) and the environment (e.g., precipitation patterns, natural disasters; Simberloff 1995, Whitlock 2000). Species with a small population size, and especially those species that naturally inhabit small geographic ranges (i.e., range-restricted species), are particularly vulnerable to these proximate causes when suitable habitat area within those ranges has been substantially reduced (Harnik et al. 2012). Such habitat reduction can further isolate metapopulations (i.e., generally isolated groups of conspecific individuals between which dispersal occasionally occurs). For many extant species particularly vulnerable to these pressures, we do not know which metapopulations are viable (i.e., likely to persist 100 years into the future) because of a lack of monitoring. Metapopulations can be defined by the 99th percentile for the focal species' dispersal distance. Namely, all conspecific territories that can be linked together via straight lines less than that distance (one line between each adjacent territory) and that do not cross known hard barriers (like large water bodies) constitute a metapopulation (Stith et al. 1996).

The Florida Scrub-jay (*Aphelocoma coerulescens*; hereafter scrub-jay), endemic to Florida, exemplifies vulnerability and exposure to each of the above factors. Even before intensive anthropogenic land transformation began in the 1800s, scrub-jay metapopulations were relatively small, non-contiguous, and isolated because of extreme sedentariness (most scrub-jays disperse less than 3 territory widths [Woolfenden and Fitzpatrick 2020], a distance of ~1.2 km at SSF [D. E. Brewer, Central Michigan University, pers. obs.]) and dependence upon fire-mediated oak-scrub confined to sandy ridges, river corridors, and coasts. It has been estimated that the range-wide scrub-jay population has declined by more than 90% since the 1800s largely because of fire suppression and anthropogenic habitat conversion (Woolfenden and Fitzpatrick 2020). Though there have been extensive studies of scrub-jay biology at Archbold Biological Station (ABS; e.g., Woolfenden and Fitzpatrick 1984) and at or in the vicinity of Merritt Island National Wildlife Refuge (MINWR; e.g., Breininger et al. 2006), relatively few studies have occurred in other parts of Florida where at least 8 additional genetically distinct scrub-jay groups occur (Coulon et al. 2008). These 10 genetic groups, which likely reflect habitat connectivity prior to European settlement, contain metapopulations that have become increasingly isolated in the past 200 years because of anthropogenic habitat conversion (Coulon et al. 2008, Woolfenden and Fitzpatrick 2020). In the early 1990s, there were 42 extant scrub-jay metapopulations (each separated by >12 km; Stith et al. 1996) and a total population

size (~4,000 family groups) that was approximately 25% less than 10 years before (Woolfenden and Fitzpatrick 2020). Scrub-jay population declines have continued since the early 1990s in urbanizing areas (e.g., Miller and Stith 2002) and even on lands managed to sustain natural resources, including scrub-jays (Boughton and Bowman 2011). Many of the 42 metapopulations present in the early 1990s are now extirpated (Coulon et al. 2008), though another intensive statewide survey is required to determine how many metapopulations and scrub-jays currently exist. One recent estimate suggests that there are currently between 6,000 and 9,000 scrub-jays remaining (~3,000 family groups; BirdLife International 2020). Management strategies such as prescribed fire and mechanical disturbance that promote an early successional, scrub-vegetation state can be used to preserve or grow scrub-jay populations (e.g., at ABS; Boughton and Bowman 2011). Such management likely would help to guard remaining metapopulations throughout the state against extirpation, though studying population trends at numerous sites beyond ABS and MINWR may help to identify where to focus conservation efforts and how those efforts should be fine-tuned based on location. To better understand scrub-jay metapopulations throughout the state, however, existing datasets that document population trends must be used effectively.

The metapopulation that includes scrub-jays at Seminole State Forest (SSF; 28.9 N, 81.5 W; metapopulation 18 in Stith et al. [1996]) in north-central Florida is >175 km to the north of ABS (27.1 N, 81.2 W) and >70 km west-northwest of MINWR (28.3 N, 80.4 W). This distance functionally precludes SSF scrub-jays from interbreeding with these well-studied scrub-jay groups (Coulon et al. 2008). Scrub-jays at SSF are genetically most similar to those at Ocala National Forest (ONF) and in metapopulations elsewhere in the western part of north central Florida, which are all a part of the largest genetic group—based on geographic coverage—identified by Coulon et al. (2008; Fig. 1). Studies at SSF could improve our understanding of scrub-jays in the understudied region wherein the property occurs and could be compared to studies completed at ABS or MINWR to determine the applicability of those studies to other metapopulations, and so optimize conservation efforts. The population of scrub-jays at SSF has been monitored since 2008 and exemplifies an opportunity to effectively use an existing dataset to better understand population trajectory. The viability of the SSF scrub-jay population could possibly be used as an indicator for the viability of the metapopulation that it prominently exists within, though monitoring of populations outside of SSF would be required to confirm this possibility. Determining scrub-jay population viability at SSF could also indicate if management practices implemented there,

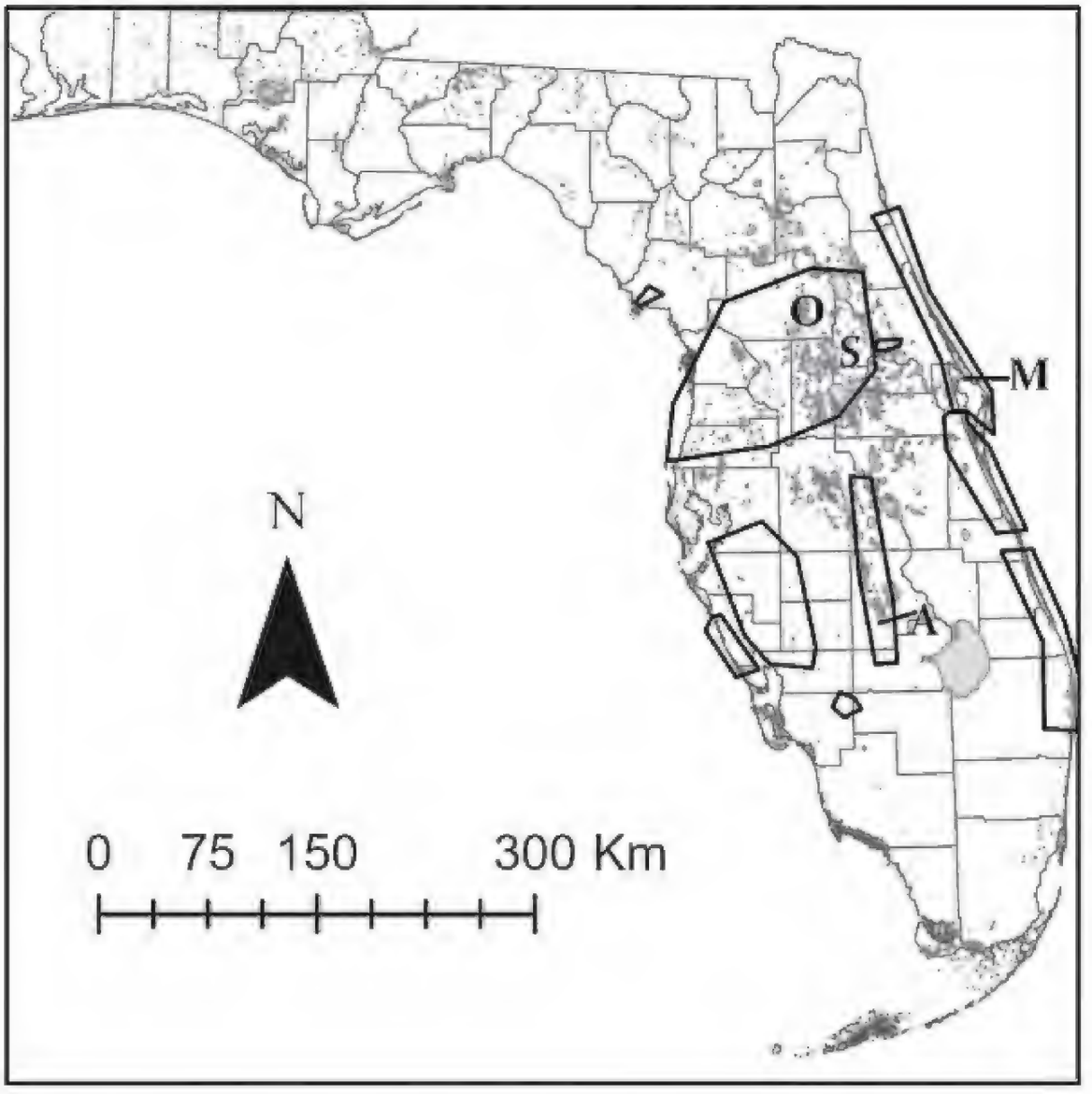

Figure 1. Sites where scrub-jays occur that are discussed herein: S = Seminole State Forest, O = Ocala National Forest, M = Merritt Island National Wildlife Refuge, and A = Archbold Biological Station. The locations of these letters (which do not correspond to polygons) indicate the center of the sites described. Major solid lines outlining polygons indicate the most biologically likely scrub-jay genetic groups as identified by Coulon et al. (2008). Minor solid lines indicate county boundaries and light gray shading indicates waterbodies.

or at locations where similar management techniques are being used, should be modified to increase the likelihood of long-term scrub-jay persistence. Additionally, demonstrating the usefulness of long-term scrub-jay population data for projecting future population change could inspire such analysis of similar datasets to help inform management decisions. Therefore, our objectives were to report SSF population survey data from 2008 to 2016 and to use only these data to estimate population viability of the scrub-jay population at SSF using a simple analysis that managers could easily implement elsewhere.

Methods

Study site.—Since being established in 1990 in Lake County, Florida, SSF has expanded to include approximately 113 km^2. Approximately 6 km^2 of this area is xeric scrub dominated by sand live oak (*Quercus geminata*), myrtle oak (*Quercus myrtifolia*), and sand pine (*Pinus clausa*) currently in suitable condition (Woolfenden and Fitzpatrick 2020) for scrub-jay occupation and reproductive success at population replacement levels (R. Risch, Florida Forest Service, pers. obs.). Other land cover types that occur on the property, among which scrub is interspersed, include upland pine forests (e.g., slash pine [*Pinus elliottii*], long leaf pine [*Pinus palustris*]) and riparian vegetation (e.g., sabal palm [*Sabal palmetto*], pond pine [*Pinus serotina*]) along Blackwater Creek and its tributaries within the Wekiva River Basin. Existing scrub vegetation is maintained via prescribed fire. The greater SSF area (including a state park and state reserve) is bordered on the east, south, and west by dense human settlement including the Orlando metropolitan area. The southeastern part of ONF borders SSF on the north and is where the nearest concentration of scrub-jays occurs, though the approximate number of scrub-jays in the southeast part of ONF is unknown.

Population surveys.—Between 2008 and 2016, population surveys occurred on average eight times per year (every six weeks) in March, April, June, July, September, October, and December. Each year, the majority of the scrub-jays present were banded with unique combinations of colored bands and trained to approach the surveyor upon hearing a whistle (80–90% of scrub-jays were typically banded at any time). The surveyor searched all suitable scrub patches at SSF (i.e., those 5–15 years post-burn) for scrub-jays during a one-week period during each population survey. Scrub patches outside of the 5–15 years post-burn range were rarely used and surveyed only if scrub-jays were known to use those sites based on daily work responsibilities of the surveyor at the property. Scrub-jays were attracted to the playback of conspecific vocalizations broadcast by the same surveyor each year. The surveyor counted both the number of all-purpose territories defended by at least one bird—these territories tended to be spatially stable through time—and the number of scrub-jays. Surveys typically lasted 2–20 minutes, depending on how quickly the surveyor identified individuals documented at the territory during previous surveys. The surveyor did not conduct surveys when aerial predators were present. During surveys, the surveyor broadcast conspecific audio from locations throughout the focal territory until they observed the focal individuals. This process also occurred at scrub patches that the experienced surveyor believed had a reasonable chance of containing scrub-jays but where recent, previous surveys had not indicated scrub-jay presence. The surveyor re-visited territories where they had previously documented scrub-jays but did not observe scrub-jays during the first survey; they re-visited the territory at least twice to confirm absence. This same process (two re-visits) occurred for missing breeding individuals, and for missing helpers, at least one re-visit occurred (the scrub-jay cooperative breeding system is described by Woolfenden and Fitzpatrick [1984]). The surveyor completed all surveys in a one-week period to minimize the risk of double-counting due to movement of un-banded individuals. Further, the scrub-jays that were present tended to approach the surveyor, which minimized the likelihood of missing individuals. We are therefore confident that our counts of scrub-jay population size at SSF are accurate. We chose to report data only from the March population survey herein, and to base our analysis on these data, because this survey was the least likely to miss un-banded individuals such as juveniles (yet to be trained to approach the surveyor), incubating or brooding individuals, or recent immigrants.

Population viability analysis (PVA).—Using the counts of scrub-jays in March between 2008 and 2016, we calculated population change (λ), which indicates how the SSF population size (N) changed from one year (t) to the next (t+1):

$$\lambda = \frac{N_{t+1}}{N_t}$$

A λ value for 2008 of 0.91, for example, indicates that between March of 2008 and March of 2009, the total number of scrub-jays at SSF declined by 9%.

We used a discrete model to predict temporal population change:

$$N_{t+1} = \lambda \times N_t$$

We used this model to project population change 100 years into the future based on random selection of a λ value for each year for 10,000 populations. We randomly selected the λ values from a log-normal distribution (the best fit for the observed λ values) that was structured based on the mean and standard deviation (SD) that we calculated using the natural log of each observed λ value from 2008 to 2015. Each simulated population began at the mean population size observed from 2008 to 2016 (107). Though we did not model environmental and demographic stochasticity explicitly, we assumed that the natural variation in λ that we simulated accounted for the majority of these factors (effects of future climate change, however, are absent).

We estimated probability of extirpation for the SSF population based on the proportion of the 10,000 simulated populations that declined to below 3 individuals during the 100-year simulation. We chose this threshold for extirpation because less than 3 individuals would be smaller than the mean observed family size at SSF during our study period (2.8) and would indicate that breeding success had likely ceased or would soon.

We also estimated how different long-term means of yearly λ values for the SSF population might affect extirpation probability and population size. To do so, we systematically increased the mean λ value from the observed 0.96 by iteratively adding 0.01, 0.02, 0.03, and 0.04 to all observed yearly (2008 to 2015) λ values, calculated the natural log for each of those sets of derived values, and then calculated the mean and standard deviation to conduct a PVA as described above. Then we determined which artificially increased mean λ values (of 0.97, 0.98, 0.99, and 1.00) resulted in at least 95% of the simulations not ending in extirpation (i.e., $N \geq 3$ after 100 years). We conducted analyses in R (v. 3.6.2; R Core Team 2019). Variability reported is standard deviation unless otherwise noted.

Results

Between 2008 and 2016, mean population size in March at SSF was 107.22 ± 16.47 (Fig. 2). The population size ranged from a high in 2008 of 137 to a low in 2015 of 90 and ended at 97 individuals in 2016 (Fig. 2). During the study period, the 8 λ values for population size had a mean of 0.96 ± 0.11 and tended to decline temporally (6 of 8 λ values indicated a decline; Fig. 3). Mean number of territories in March was 38.56 ± 4.82 (Figs. 2, 4) and the 8 associated λ values for territories had a mean of 1.0 ± 0.16 (4 indicated a decline; Fig. 3). The λ values for number of scrub-jays and number of territories did not tend to vary synchronously with respect to degree or direction of change (Fig. 3). Mean scrub-jays per territory in March was 2.79 ± 0.39, tended to decline temporally (5 of 8 transitions), and also did not tend to vary in the same direction as number of territories through time (Fig. 4). We did not detect a significant correlation between number of territories and number of scrub-jays per territory ($r = -0.288$, $df = 7$, $P = 0.45$), though sample size and power were low such that an undetected negative relationship is possible.

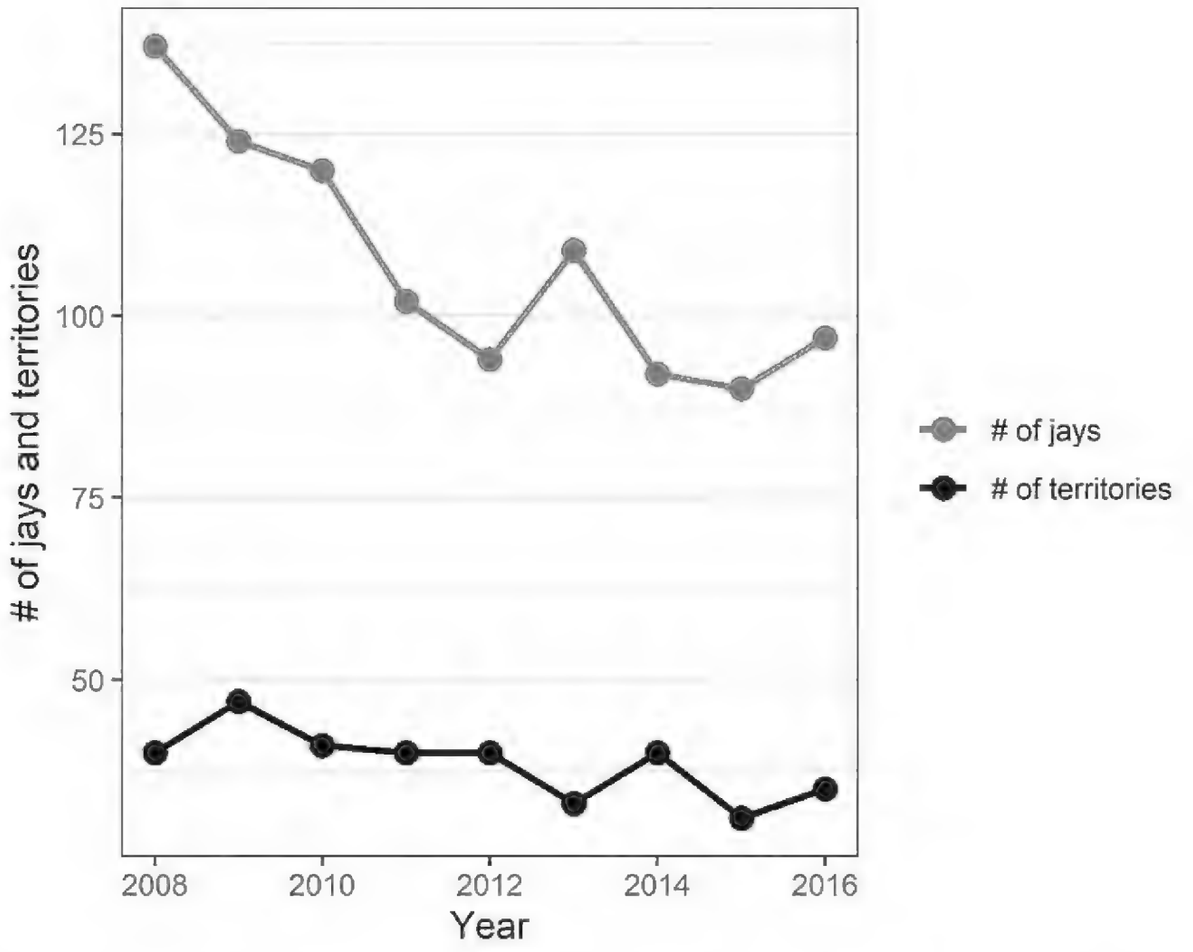

Figure 2. Variation in number (#) of scrub-jays (gray) and number of territories (black) at Seminole State Forest between 2008 and 2016 during the March population survey.

The range of λ values observed between 2008 and 2015 resulted in an extirpation probability within 100 years of 0.78 (Fig. 5). Upon systematically increasing the mean of yearly λ values for our study period, we found that mean λ values of 0.97, 0.98, 0.99, and 1.00 resulted in extirpation probabilities of 0.44, 0.14, 0.02, and 0.00, respectively (Fig. 5). With increasing λ, projected final populations sizes tended to increase and median year of extirpation tended to occur later (Table 1).

Discussion

Based on the generally declining number of scrub-jays that occurred at SSF between 2008 and 2016 (Figs. 2, 3), our PVA indicated that the SSF population has a 0.78 extirpation probability within the next 100 years (Fig. 5) if the range of λ levels that we observed is representative of future conditions. The number of territories at SSF were relatively constant during the study period (Figs. 2, 4) compared to the period between 1992 and 2010 when the number of territories more than doubled (Boughton and Bowman 2011). Both number of

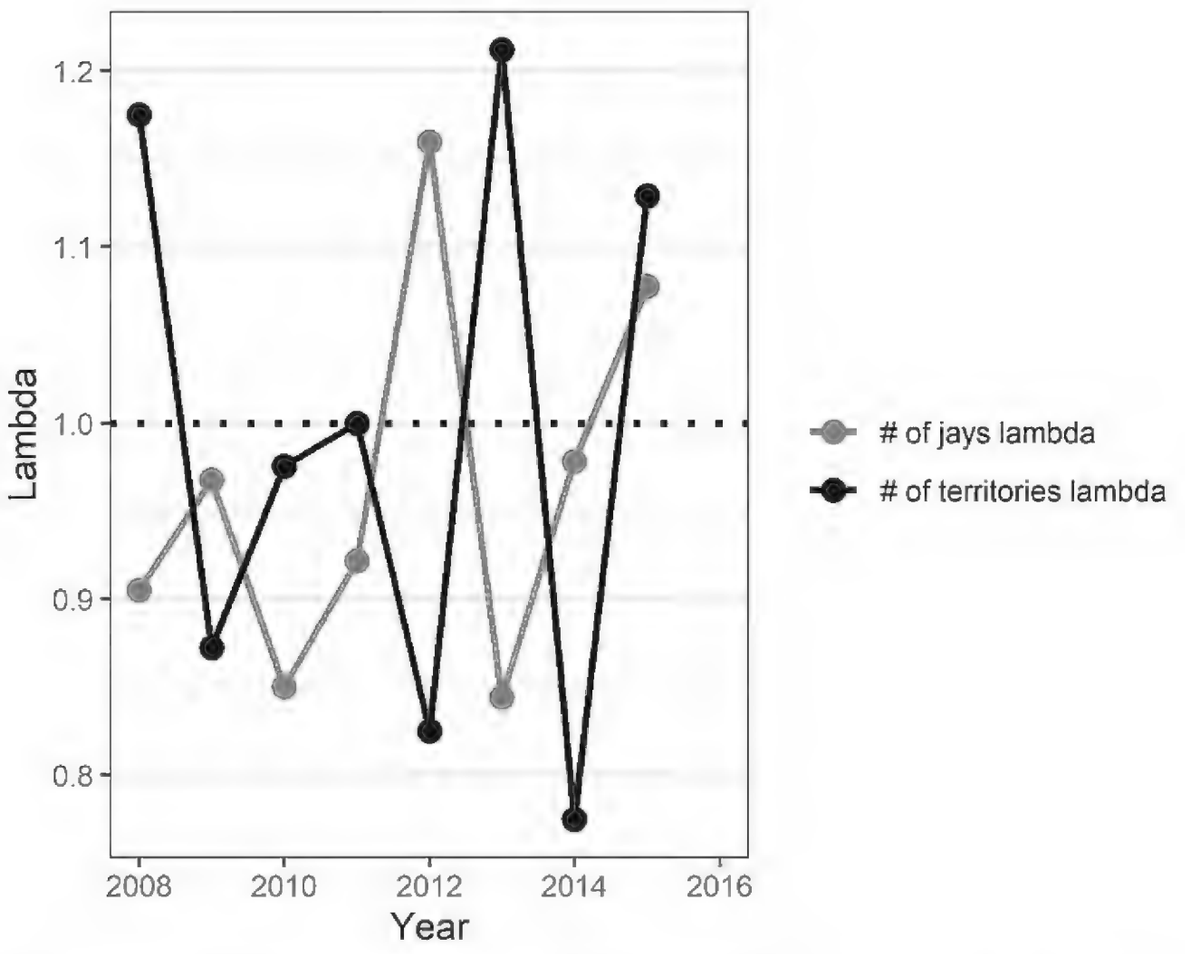

Figure 3. Variation in population change (lambda) values for number (#) of scrub-jays (gray) and number of territories (black) at Seminole State Forest between 2008 and 2015, based on March population surveys. The horizontal dashed line represents no growth. The lambda values for number of scrub-jays and for number of territories tended to vary in the opposite direction.

territories and number of scrub-jays per territory did, however, tend to decline during the study period and generally varied between years in opposite directions (Fig. 4). We also observed that between-year changes in total scrub-jays and territory numbers at SSF often varied in opposite directions (Figs. 2, 3). The cooperative breeding system of scrub-jays (Woolfenden and Fitzpatrick 1984), in which some juveniles remain helpers on parental territories, could explain both of these patterns (Figs. 2, 3, 4). Fewer scrub-jays per territory would be expected when there are more territories resulting from dispersal and vice versa (Fig. 4). Opposite trajectories in overall scrub-jay population and territory number (Figs. 2, 3) could indicate an oscillation between productive years when habitat is saturated (juveniles survive because of favorable conditions but do not disperse because of lack of suitable habitat available) and mortality-mediated habitat availability, which facilitates territory establishment by former helpers. Productive years and saturated habitat might result in larger territories with more

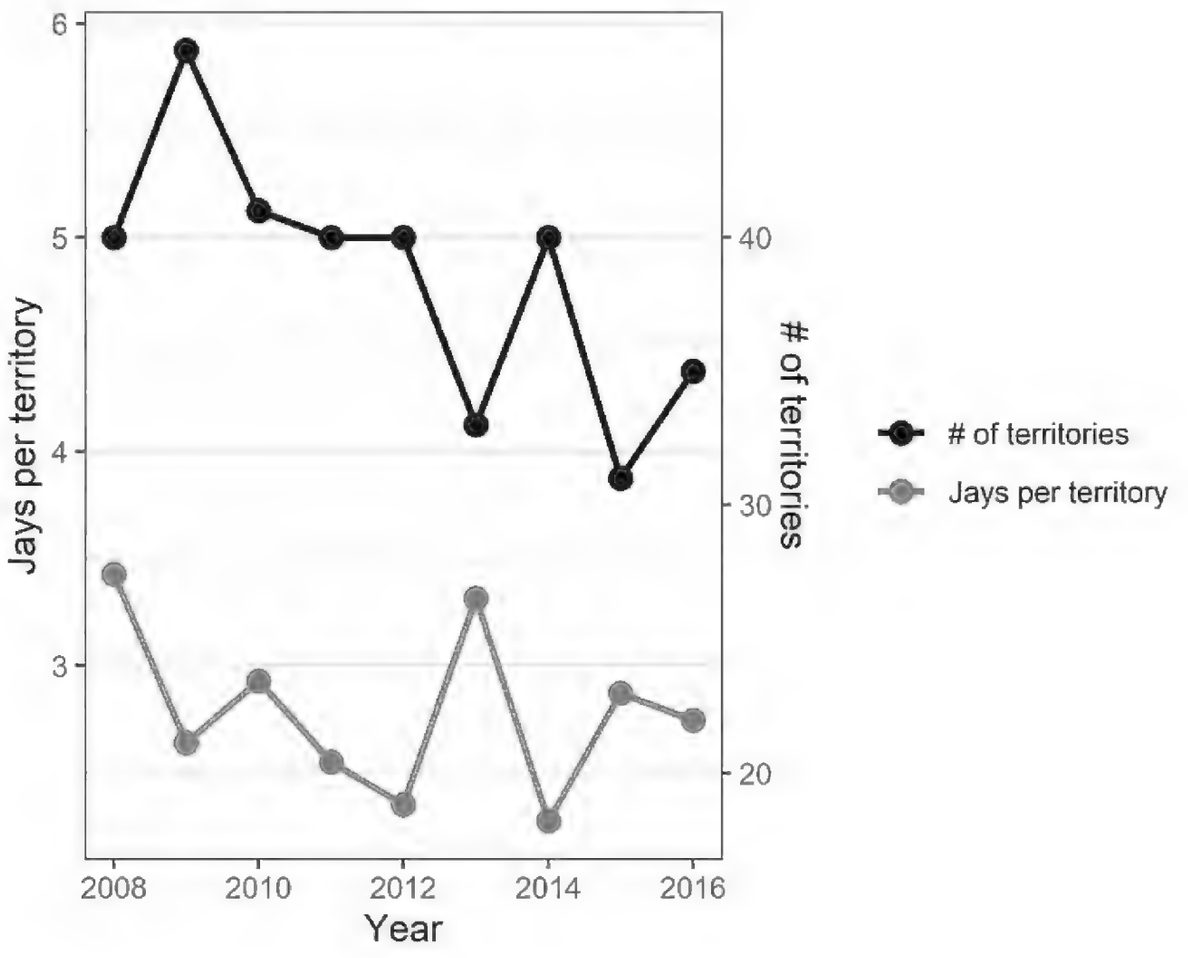

Figure 4. Mean number of scrub-jays per territory at Seminole State Forest (gray; left *y* axis) and number of territories (black; right *y* axis) during March population surveys from 2008 to 2016. The number of territories and number of scrub-jays per territory tended to vary in the opposite direction.

scrub-jays per territory (and more scrub-jays overall, fewer territories), whereas habitat vacated by deceased breeders might result in colonization by former helpers that establish smaller, less scrub-jay-dense territories (and fewer scrub-jays overall, more territories).

Regardless, it is clear that the SSF scrub-jay population, and perhaps the metapopulation to which it belongs, requires modified management action to improve conditions to attain a reasonably low extirpation probability (e.g., < 0.05, acceptable risk) within the next 100 years. An additional 18 km^2, approximately, of xeric-scrub could be restored at SSF as habitat for scrub-jays. Even moderate expansion of area managed as scrub-jay habitat within SSF boundaries could possibly, if targeted around patches with high scrub-jay densities (Breininger et al. 2006), be sufficient to increase the mean λ value on the property from 0.96 to 0.99 or greater and so significantly reduce extirpation probability (Fig. 5). Our simulations suggest that such a modest increase in λ could reduce extirpation risk to acceptable

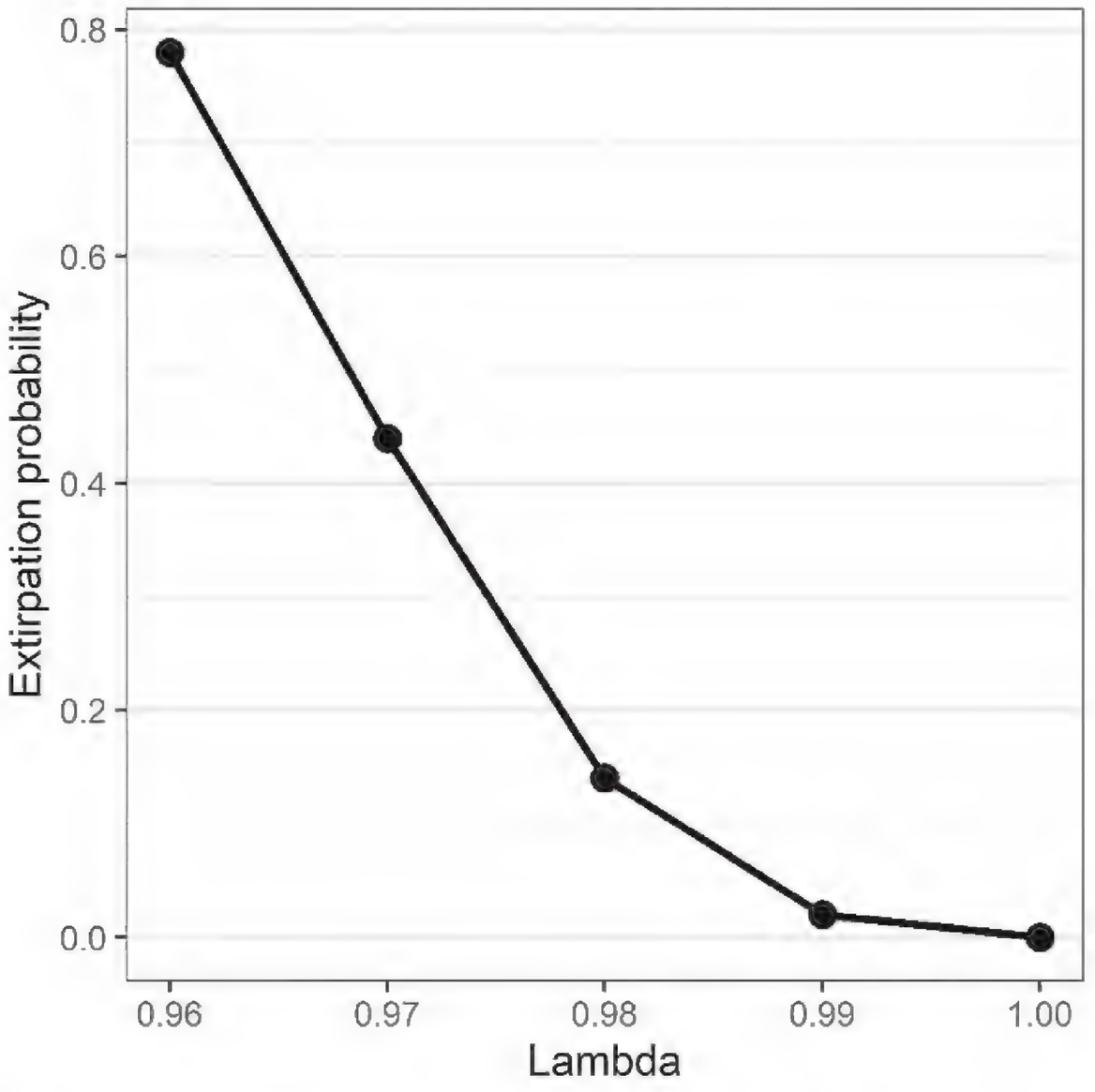

Figure 5. Projected extirpation probabilities during a 100-year period as a function of different mean values of population change (lambda) for the Seminole State Forest scrub-jay population. The mean lambda value observed between 2008 and 2015 was 0.96 and extirpation probabilities are based on the proportion of 10,000 simulations that resulted in extirpation.

levels at SSF and perhaps for the entire metapopulation, given that most of its scrub-jays are assumed to exist in SSF. Further, modifying the frequency of prescribed fire or mechanical removal of larger trees in existing scrub could improve the quality of habitat and so increase scrub-jay numbers (Breininger et al. 2014, Woolfenden and Fitzpatrick 2020). We did not conduct habitat surveys during this study; therefore, we do not know the primary driver of population change at SSF. Regardless, our results suggest that conservation funds meant to benefit scrub-jays would, in conjunction with habitat surveys, likely be well-spent on conducting management activities at SSF.

Table 1. Mean final scrub-jay population size and standard error (SE), median years until extirpation, and years until first extirpation based on 10,000 simulations for different mean values of population change (λ). Each simulation had a duration of 100 years. The mean λ value observed at Seminole State Forest between 2008 and 2015 was 0.96.

λ	Mean final population size (SE)	Median years until extirpation	Years until first extirpation
0.96	1.67 (0.04)	75	28
0.97	6.44 (0.11)	85	37
0.98	20.02 (0.31)	88	41
0.99	55.26 (0.78)	89	49
1.00	150 (2.13)	89	65

A limitation of our PVA is that it did not explicitly consider environmental or demographic stochasticity or density dependence but rather relied on the variation in observed λ values to encapsulate such processes. Regarding catastrophes, we believe that our approach is justifiable given that SSF is distant from the coast, which causes the catastrophic effects of hurricanes to generally be avoided. Epidemics, however, can substantially increase extirpation probability (Breininger et al. 1999) and exemplify a stochastic event that may not have occurred during our study period that could occur in the future. Similarly, wildfires could also occur and may increase in frequency and severity because of climate change. We believe, however, that the simplicity of our model is advantageous in that only population survey data are required, which, in many cases, is all that managers who wish to determine the trajectory of their population possess. As with all PVAs (Lacy 2019), but especially with respect to ours because of its simplicity, caution should be used when interpreting our results which are meant as a rough guide rather than an exact prediction about how SSF scrub-jay population size may change in the future. If scrub-jays are monitored for longer periods of time, the accuracy of analyses such as ours will likely improve.

Given that many of the remaining scrub-jay metapopulations are currently likely declining (Coulon et al. 2008, Boughton and Bowman 2011), it is crucial to identify which of these metapopulations could feasibly be conserved, which metapopulations require management modification, and which metapopulations could benefit from habitat acquisition. Monitoring efforts, such as Audubon Florida's Jay Watch community science (i.e., citizen science) program, can produce accurate population survey data (Miller et al. 2015) and could be expanded to other sites throughout Florida. These or similar data could be used to complete periodic viability analyses, as demonstrated by the current study, for each remaining metapopulation to help inform decision

makers about where limited conservation resources would be most effectively allocated. Though he used a more complex (individual-based, spatially explicit) viability analysis than the one described herein, Stith (1999) completed an analysis of metapopulation viability previously. Another such effort could help to identify areas of current conservation need and also help managers to identify which scrub-jay populations are being managed in a way that can sustain scrub-jays and should be emulated. Our approach could feasibly be incorporated into a state-wide metapopulation analysis by providing a simple process by which to use site-specific scrub-jay population trend data that might otherwise remain largely unused. It will be important, however, to consider not just changes in λ but also to identify what factors (e.g., habitat management, genetic diversity) are influencing that metric of population change.

The Florida Scrub Working Group approach currently being employed helps to facilitate the communication required to achieve such state-wide goals. When possible, those seeking advice about scrub-jay habitat management techniques should prioritize consulting land managers at properties where scrub-jays within the same metapopulation (Stith et al. 1996), or at least the same genetic group (Coulon et al. 2008), have been successfully managed. For example, managers of scrub-jay habitat in the Northeast Florida Scrub Working Group region may benefit more from communicating with the managers at SSF than with managers in other parts of the state given regional variation in scrub types and re-growth rates. This collaborative, biologically informed approach among managers based on which metapopulation or genetic group is being managed, rather than being based solely on which species is being managed, could both improve our ability to conserve scrub-jays and serve as a model for conserving other sedentary, range-restricted species.

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SEASONAL AND ANNUAL ABUNDANCE AND DISTRIBUTION OF THE AVIAN COMMUNITY IN HUGUENOT MEMORIAL PARK, FLORIDA, FROM 2017 TO 2020

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Abstract.—We compared annual and seasonal abundances of the avian community among land cover types at Huguenot Memorial Park (MP), Jacksonville, northeastern Florida, from January 2017 to August 2020. Land cover types included the coastal scrub, banks of the St. Johns River, intertidal marsh of Ft. George Inlet, beach facing the Atlantic Ocean, peninsula that extends from the beach, and dunes located between the inlet and ocean. Of the 129 surveys conducted, we recorded 155 taxa, of which eight species were state or federally listed birds. We recorded the greatest richness in the inlet area and the greatest abundance in the peninsula. Total abundances were greater in spring and summer than autumn and winter. Royal Tern (*Sterna maxima*), Laughing Gull (*Larus atricilla*), and Brown Pelican (*Pelecanus occidentalis*) were the most common species in the park and their offspring contributed to the greater numbers recorded in spring and summer in the peninsula and dunes. Taxon richness was greatest in 2019 and fewest in 2018. The hurricanes of 2017 and 2018 may have contributed to lower richness in the coastal scrub-riverside and inlet areas and sand redistribution across the dunes and peninsula may have contributed to increased counts of Red Knot (*Calidris canutus*) and Least Tern (*Sternula antillarum*) in the latter years. Local and regional monitoring of prey availability, predator effects, and habitat quality of coastal areas in conjunction with avian population surveys would increase understanding of the impacts of extreme storm events, sea level rise, and coastal development on roosting, foraging, and breeding activities.

Key words: annual, abundance, avian, Florida, habitat, monitoring, seasonal

Given the pressures of urban development, recreational-user demand, and climate change, temporal analysis of habitat use by avian populations is critical to informing resource managers of conservation priorities and actions (Epstein 2019). Huguenot Memorial Park (MP), Jacksonville, Florida, is part of the Florida Birding Trail and includes Florida Fish and Wildlife Conservation Commission's (FWC) Ft. George Inlet Critical Wildlife Area (CWA; FWC 2019), which is used by roosting and nesting seabirds and shorebirds. The diversity and abundance of shorebirds and seabirds are likely related, in part, to the variety of coastal land cover types in the area, which include salt marshes and mud flats along the Ft. George Inlet, coastal dune

and swale complexes, sandy beaches facing the Atlantic Ocean, sandy shores along the St. Johns River, and coastal scrub upland of the river (England-Thims & Miller, Inc. 2008). These land cover types provide resources for migrating shorebirds, seabirds, and passerines. For example, the shorelines of the inlet and beach are used for foraging and roosting activities by migratory and resident seabirds and shorebirds, and the peninsula's CWA provides nesting habitat for seabirds (Royal Tern [*Sterna maxima*], Laughing Gull [*Larus atricilla*], and Brown Pelican [*Pelecanus occidentalis*]) during the spring and summer months (England-Thims & Miller, Inc. 2008).

Because the park faces the St. Johns River, Ft. George Inlet, and Atlantic Ocean, Huguenot MP is especially vulnerable to habitat loss and degradation from erosion and flooding by extreme storm events. In September and October 2016, erosion caused significant damage along the road adjacent to the St. Johns River, which resulted in closure of Huguenot MP from October 2016 to March 2017. Rainfall and storm surge from Hurricanes Irma and Dorian in 2017 and 2018, respectively, contributed to flooding of the coastal areas of the park. Convertino et al. (2011) suggested that shorebird diversity may benefit from extreme storm events that can increase tidal flat areas for foraging and nesting activities. No previous study has investigated the impact of extreme storms on avian diversity on the northeastern Florida coast.

The objectives of the study were to describe avian species abundance and richness patterns in the annual and seasonal use of land cover types in Huguenot MP from 2017 to August 2020, with a focus on shorebirds and seabirds. Following the Shorebird Management Plan (England-Thims & Miller, Inc. 2008), we conducted weekly bird surveys along the combined coastal scrub community and the banks of the St. Johns River, dunes, peninsula, beach, and tidal marsh of Ft. George Inlet. Results from our study contribute to understanding of variability of avian population with successive storms in Florida.

Methods

We compared temporal patterns in avian abundance (number of individuals per survey) and richness among five locations surveyed at Huguenot MP (Fig. 1). The combined length of the coastal scrub and banks of the St. Johns River was 1.9 km and the length of the marsh and mud flats of the Fort George Inlet was 2.0 km. The peninsula was composed of sand dunes and the sandy beach areas were 1.8 km in length (Fig. 1). We conducted surveys every week during the breeding season from March to August, and then every other week from September to February. One to three surveyors would walk between the coastal scrub and river, to the beach, and then along the peninsula, dunes, and Ft. George Inlet, recording counts per species (Fig. 1). Surveys would begin between 0800 to 0900 and take two to five hours. We used binoculars and spotting scopes to aid in bird identifications. We classified birds as adults, nestlings with downy feathers, fledglings, and juveniles. Because of the relatively abundant numbers of nesting Royal Terns, we counted a proportion of the population and then multiplied by the entire dune area.

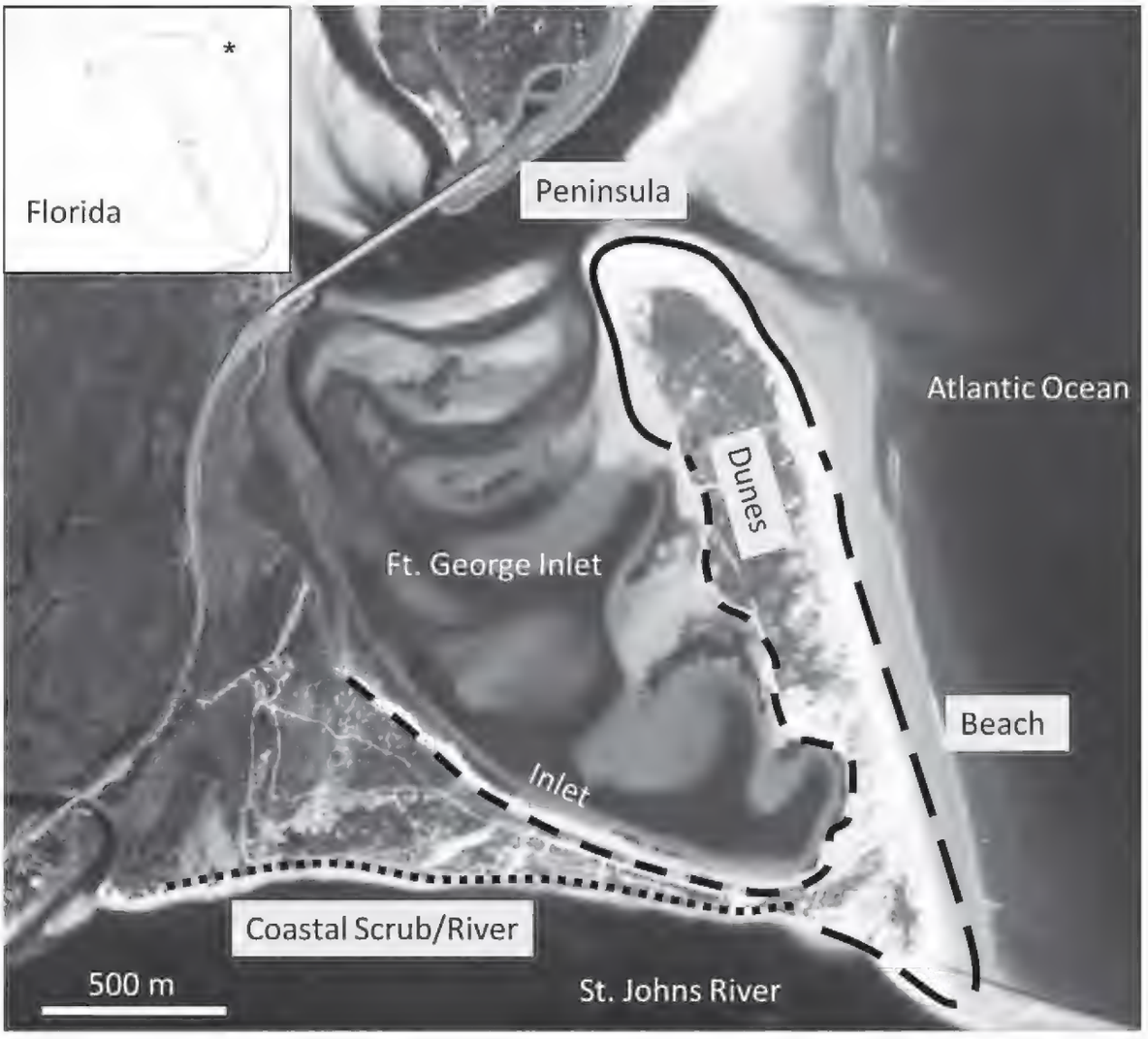

Figure 1. Location of land cover types surveyed in Huguenot Memorial Park, Florida from 2017 to 2020.

We compared seasonal (winter: December–February, spring: March–May, summer: June–August, and autumn: September–November; Sprandel 2007) and annual differences for common birds that were recorded in >50% of the surveys sampled throughout the study period. We combined numbers of nestlings, fledglings, and juveniles per species per survey for analyses. Because abundance and richness data did not meet the assumptions of parametric statistics, we used Kruskal-Wallis tests and Dunn pairwise comparisons (alpha = 0.05; Minitab version 19, Minitab, Inc., State College, PA). We report means and standard errors.

Results

From January 2017 to August 2020, we recorded 155 avian taxa from six orders (n = 129 surveys; Table 1). Of the 24 common taxa (in >50% of the 129 surveys), we recorded Laughing Gull, Brown Pelican, and Royal Tern in >80% of the surveys. Eight state or federally listed birds were present in the park throughout the study period (Tables 1 and 2). Eight species had chicks hatched in Huguenot MP with early life stages categorized as nestling (N), fledgling (F), or flight-capable

Table 1. Percent frequency of occurrence and mean (± SE) counts per survey within each land cover type of common taxa at Huguenot Memorial Park, Florida, from 2017 to 2020 (n = 129 surveys). We indicate significant seasonal (S) or annual (A) differences in mean counts per survey (P < 0.05).

Taxa	Scrub-river	Inlet	Dunes	Peninsula	Beach
Accipitriformes					
Osprey (*Pandion haliaetus*)- S, A	56%, 2.3 ± 0.9	65%, 2.7 ± 0.3		43%, 2.1 ± 0.3	36%, 2.0 ± 0.2
Charadriiformes					
Black Skimmer- S	3%, 15.8 ± 6.7	32%, 44.8 ± 10.3	5%, 2.8 ± 2.3	62%, 79.6 ± 12.5	41%, 28.8 ± 10.8
Black-bellied Plover- S	3%, 1.4 ± 0.6	76%, 14.2 ± 1.5		45%, 11.5 ± 1.9	27%, 11.0 ± 1.9
Dunlin- S	2%, 3.0 ± 2.0	50%, 136.6 ± 25.1		35%, 141.7, ± 49.0	9%, 202.5 ± 121.1
Herring Gull- S	8%, 20.3 ± 7.7	8%, 3.4 ± 2.0	1%, 1.0	68%, 10.9 ± 3.7	73%, 19.8 ± 8.6
Laughing Gull- S	56%, 19.5 ± 4.4	85%, 180.4 ± 33.8	53%, 2,562.6 ± 208.7	90%, 1,850.2 ± 193.6	93%, 389.5 ± 40.5
Ring-billed Gull	8%, 1.2 ± 0.1	24%, 5.1 ± 1.2	1%, 1.0	56%, 9.0 ± 1.7	72%, 12.1 ± 3.6
Royal Tern- S	26%, 34.4 ± 17.1	47%, 17.0 ± 7.3	44%, 1,847.2 ± 259.3	87%, 834.9 ± 111.1	92%, 331.4 ± 32.9
Ruddy Turnstone- S	5%, 9.1 ± 5.4	68%, 8.6 ± 0.9	2%, 2.5 ± 1.5	69%, 10.0 ± 1.2	69%, 16.9 ± 2.3
Sanderling (*Calidris alba*)	1%, 5.0	44%, 15.5 ± 3.4		67%, 26.3 ± 3.5	71%, 31.8 ± 3.8
Sandwich Tern- S	5%, 2.2 ± 0.8	17%, 11.4 ± 6.4	5%, 1.7 ± 0.4	49%, 17.6 ± 4.2	54%, 46.6 ± 9.6
Semipalmated Plover- S	1%, 1.0	50%, 141.1 ± 25.5		42%, 172.0 ± 31.8	25%, 77.0 ± 24.9
Short-billed Dowitcher- A		58%, 53.2 ± 10.3			3%, 15.3 ± 9.9
Willet- S, A	8%, 1.8 ± 0.4	67%, 6.9 ± 0.6	1%, 2.0	36%, 3.2 ± 0.8	54%, 2.8 ± 0.3
Wilson's Plover- S, A	7%, 5.9 ± 2.7	54%, 10.3 ± 1.4	2%, 1.0 ± 0.0	27%, 2.7 ± 1.4	18%, 3.1 ± 0.5
Columbiformes					
Mourning Dove (*Zenaida macroura*)- A	62%, 5.0 ± 0.7	23%, 4.0 ± 0.7	2%, 3.0 ± 2.0	11%, 4.6 ± 0.9	2%, 1.3 ± 0.3
Passeriformes					
Boat-tailed Grackle (*Quiscalus major*)- S	56%, 7.1 ± 0.8	62%, 7.4 ± 1.0	22%, 2.5 ± 0.3	41%, 3.1 ± 0.4	59%, 7.1 ± 0.7
Northern Cardinal (*Cardinalis cardinalis*)- A	50%, 2.5 ± 0.2	2%, 2.0 ± 1.0			

Table 1. (Continued) Percent frequency of occurrence and mean (± SE) counts per survey within each land cover type of common taxa at Huguenot Memorial Park, Florida, from 2017 to 2020 (n = 129 surveys). We indicate significant seasonal (S) or annual (A) differences in mean counts per survey (P < 0.05).

Taxa	Scrub-river	Inlet	Dunes	Peninsula	Beach
Northern Mockingbird (*Mimus polyglottos*)- A	89%, 5.7 ± 0.4	13%, 1.8 ± 0.3			
Pelecaniformes					
Brown Pelican- S	20%, 22.9 ± 9.0	77%, 8.6 ± 1.1	40%, 111.0 ± 8.0	81%, 55.8 ± 7.3	81%, 35.6 ± 5.3
Great Blue Heron (*Ardea herodias*)	23%, 1.3 ± 0.3	60%, 1.9 ± 0.1	5%, 1.2 ± 0.2	20%, 1.1 ± 0.1	11%, 1.6 ± 0.3
Great Egret- S, A	35%, 2.6 ± 0.6	68%, 5.9 ± 0.6	1%, 2.0	17%, 2.0 ± 0.3	10%, 1.8 ± 0.3
Snowy Egret (*Egretta thula*)	16%, 2.7 ± 0.7	57%, 5.8 ± 0.9		11%, 1.3 ± 0.2	29%, 1.4 ± 0.1
Suliformes					
Double-crested Cormorant- S, A	9%, 4.2 ± 1.6	54%, 11.4 ± 2.0		14%, 4.7 ± 1.9	<1%, 20.4 ± 7.6
Totals					
Counts per survey- S	62.8 ± 8.9	467.4 ± 42.9	2,233.7 ± 268.7	2,741.7 ± 235.6	900.3 ± 64.4
Species richness per survey- A	10.3 ± 0.4	16.8 ± 0.5	2.0 ± 0.2	13.8 ± 0.4	12.9 ± 0.4

Table 2. Frequency of occurrence and mean (± SE) annual abundance of Florida (St) and federally (F) threatened species at Huguenot Memorial Park, Florida from 2017 to 2020 (n = 129 surveys). We indicate land cover types present per species as coastal scrub-riverside (S-R), Inlet (I), dune (D), peninsula (P), and beach (B).

Taxa	Land cover type	2017	2018	2019	2020
Charadriiformes					
American Oystercatcher- St	S-R I, D, P, B	70%, 3.6 ± 0.3	58%, 3.0 ± 0.3	78%, 4.8 ± 0.4	73%, 5.6 ± 0.6
Least Tern- St	S-R, I, P, B	53%, 10.4 ± 2.7	33%, 6.0 ± 2.5	49%, 6.1 ± 1.3	59%, 28.8 ± 9.0
Red Knot- F	I, P	27%, 33.0 ± 10.2	20%, 4.8 ± 1.4	19%, 27.7 ± 12.4	
Piping Plover- F	I, P, B	37%, 6.5 ± 1.0	25%, 2.8 ± 7.7	46%, 3.3 ± 0.5	23%, 2.6 ± 1.2
Ciconiiformes					
Wood Stork [*Mycteria americana*]- St	S-R, I, P, B	13%, 2.0 ± 0.5	13%, 1.6 ± 0.4	14%, 2.3 ± 0.5	14%, 1.3 ± 0.3
Pelecaniformes					
Little Blue Heron- St	I	3%, 1.0	3%, 1.0		
Reddish Egret- St	I, P, B	37%, 2.5 ± 0.4	53%, 3.9 ± 0.6	35%, 4.0 ± 0.7	41%, 1.6 ± 0.3
Tricolored Heron- St	I, P, B	27%, 12.0 ± 0.3	5%, 1.0	24%, 1.3 ± 0.2	9%, 1.0

juveniles (J): Laughing Gull (N, F, and J), Brown Pelican (N, F, and J), American Oystercatcher (*Haematopus palliates*; N and J), Royal Tern (N, F, and J), Sandwich Tern (*Sterna sandvicensis*; N, F, and J), Least Tern (*Sternula antillarum*; N), Willet (*Catoptrophorus semipalmatus*; J), and Wilson's Plover (*Charadrius wilsonia*; F and J; Table 3). We also observed Great Black-backed Gull (*Larus marinus*), Black Skimmer (*Rynchops niger*), Herring Gull (*Larus argentatus*), Lesser Black-backed Gull (*Larus fuscus*), and Caspian Tern (*Sterna caspia*) juveniles. We recorded six species with nestlings, fledglings or juvenile life stages in the inlet: five at the beach, five on the peninsula, and three on the dunes (Table 3).

Total avian abundances per survey differed seasonally ($H = 52.60$, $P < 0.001$) and not annually in the park ($H = 4.16$, $P = 0.244$; Figs. 2 and 3). Of the 24 common species, relative abundance of 16 species differed among seasons ($P < 0.05$), and nine were different among years surveyed ($P < 0.05$; Table 1). We recorded greater counts per survey in spring and summer (8,386.5 ± 727.6 counts per survey) when chicks were present than in autumn and winter (3,030.8 ± 691.6 counts per survey; Fig. 2). In contrast, species richness differed annually ($H = 29.8$, $P < 0.001$; Fig. 3) and not seasonally ($H = 5.7$, $P = 0.124$; Fig. 2). We recorded the greatest mean (± SE) richness in 2019 (36.8 ± 0.9 taxa per survey) and fewest in 2018 (28.8 ± 1.0 taxa per survey).

Bird abundances and species richness differed among land cover types ($H_{abundance} = 234.85$, $H_{richness} = 351.93$, $P < 0.001$) throughout the study. We recorded the greatest mean counts in the peninsula (2,741.7 ± 235.6 counts per survey) and fewest in the coastal scrub-riverside (62.8 ± 8.9 counts per survey; Table 1). By comparison, mean species richness was greatest in the inlet (10.4 ± 0.4 species per survey) and lowest in the dunes (2.0 ± 0.2 species per survey; Table 1).

The dunes, peninsula, and beach were associated primarily with seabirds. Seasonal abundances per survey were significantly different in the dunes and peninsula ($H_{dunes} = 62.73$, $H_{peninsula} = 39.55$, $P < 0.001$) and seasonal richness was significantly different in the dunes, peninsula, and beaches ($H_{dunes} = 60.82$, $H_{peninsula} = 10.59$, $H_{beach} = 24.01$, $P < 0.043$; Fig. 2). Seasonal abundance patterns were driven by the Laughing Gull, Ring-billed Gull (*Larus delawarensis*), Herring Gull, Brown Pelican, Royal Tern, Sandwich Tern, and Black Skimmer (Fig. 4). Laughing Gull counts were greater in spring and summer as compared to autumn and winter, and Royal Tern and Brown Pelican counts were greater in spring through autumn ($P < 0.007$). We recorded the greatest combined numbers of nestlings, fledglings, and juveniles in summer and fewest numbers in winter months. Laughing Gull and Royal Terns had the greatest combined numbers of nestlings, fledglings, and juveniles at Huguenot MP, representing on average,

Table 3. Land cover types (coastal scrub-riverside [S-R], Inlet [I], dune [D], peninsula [P], and beach [B]) of the surveys that had nestlings, fledglings, or juveniles present at Huguenot Memorial Park, Florida, from 20017 to 2019 (n = 129 surveys). We indicate years and seasons with nestlings, fledglings, or juveniles present with an x and the seasons with greatest numbers with an X.

Taxa	Number of surveys with nestlings, fledglings, or juveniles present	2017	2018	2019	2020	Autumn	Winter	Spring	Summer
Charadriiformes									
American Oystercatcher- I, B	2		x		x				X
Black Skimmer- I, P, B	8		x	x		X			x
Caspian Tern- P	1			x		X			
Great Black-backed Gull- B	11	x	x			X	X		
Herring Gull- B	5	x	x			X	X		
Laughing Gull- D, P	50	x	x	x	x	x		x	X
Least Tern- I	2				x				X
Lesser Black-backed Gull- I, P	12	x	x			X		x	X
Royal Tern- D, P	50	x	x	x	x	x		x	X
Sandwich Tern- B	24	x	x	x	x	x		X	X
Willet- I	6			x	x				X
Wilson's Plover- I	2				x				X
Pelecaniformes									
Brown Pelican- D	30		x	x	x	x	x	x	X
Total number of species		6	9	7	8	9	3	5	10

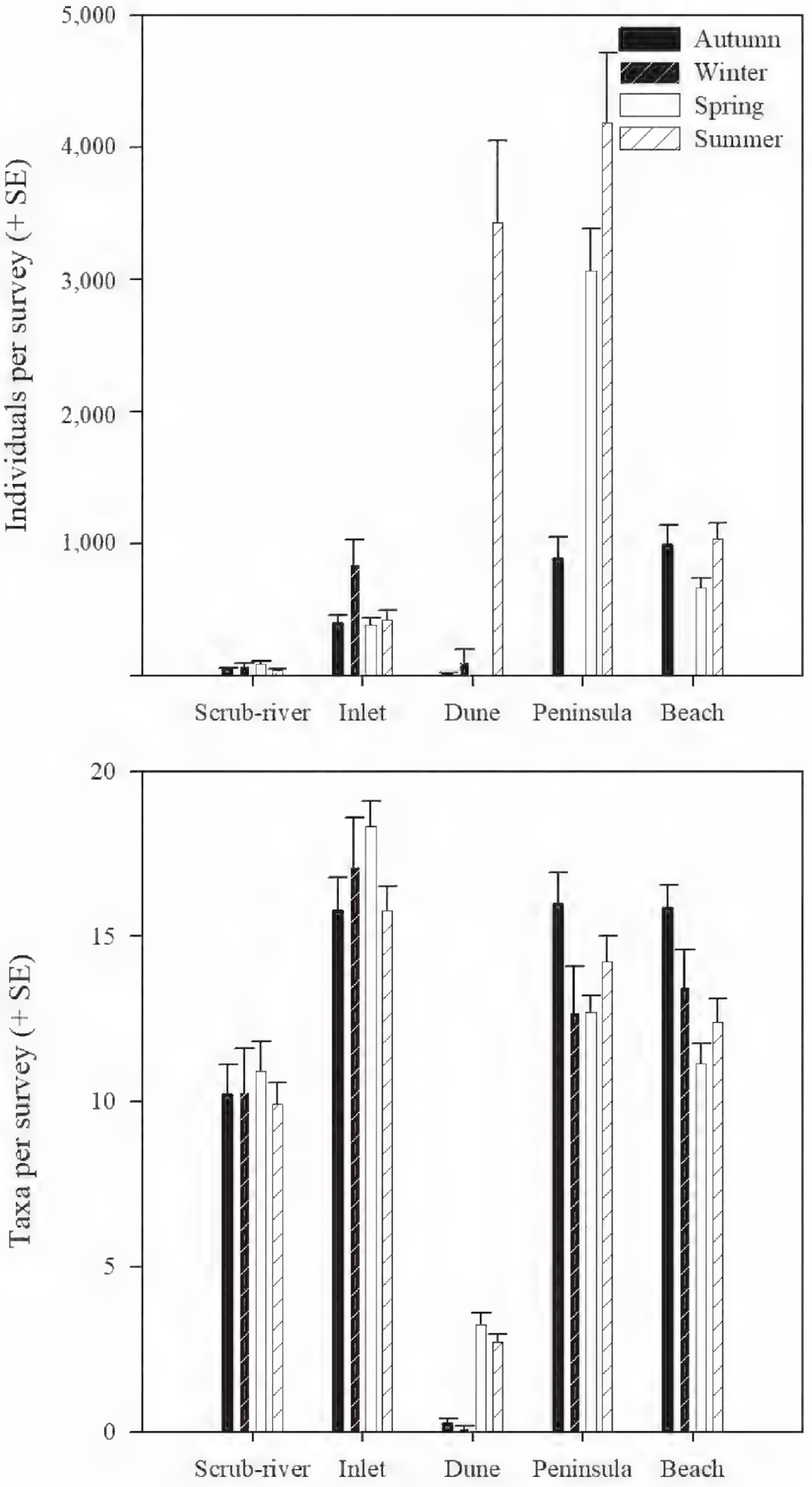

Figure 2. Seasonal mean (+ SE) bird abundance (top) and taxon richness (bottom) among land cover types surveyed in Huguenot Memorial Park, Florida, from 2017 to 2020 (n = 129 surveys).

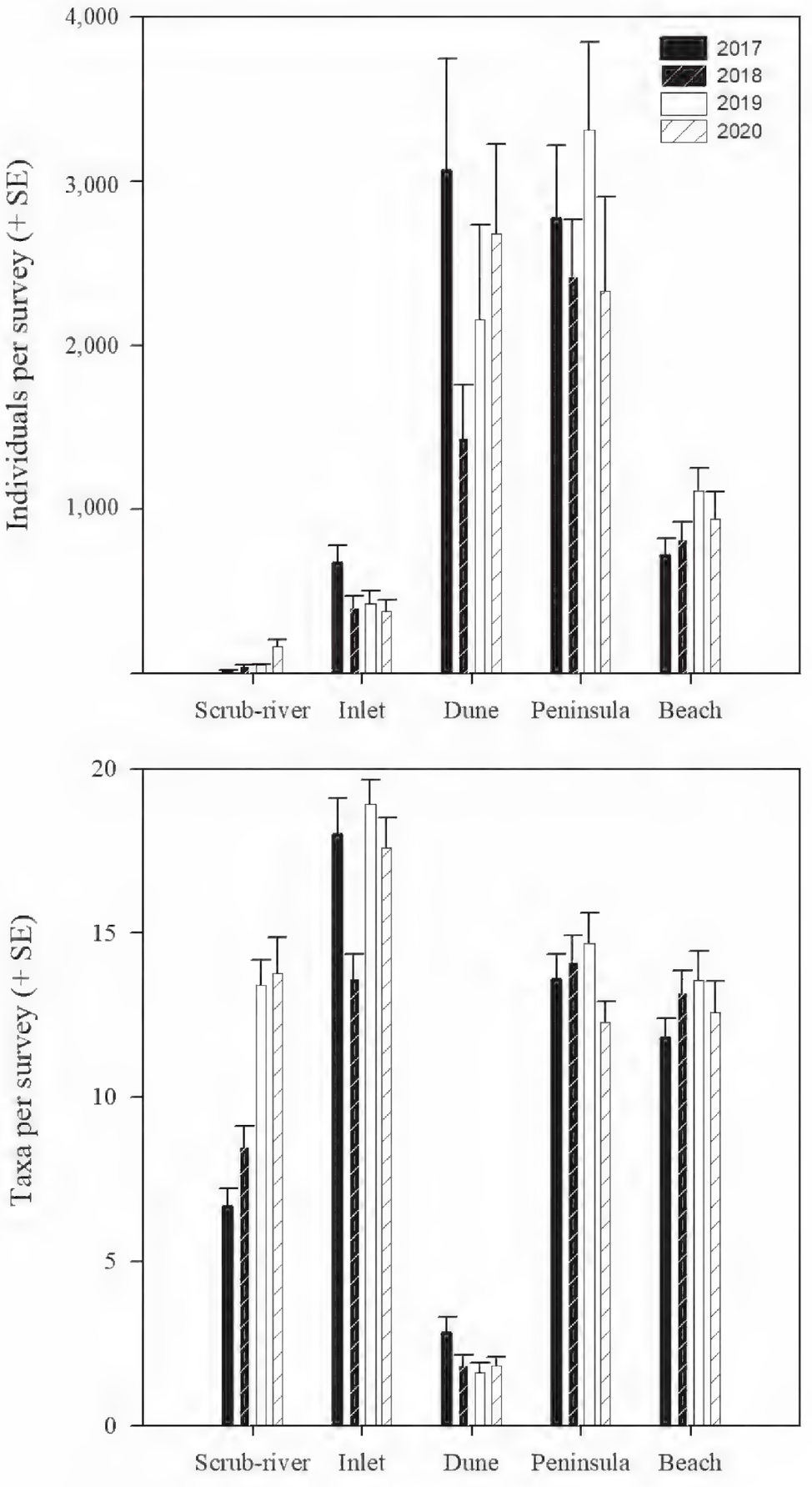

Figure 3. Annual mean (+ SE) bird abundance (top) and taxon richness (bottom) among land cover type surveyed in Huguenot Memorial Park, Florida, from 2017 to 2020 (n = 129 surveys).

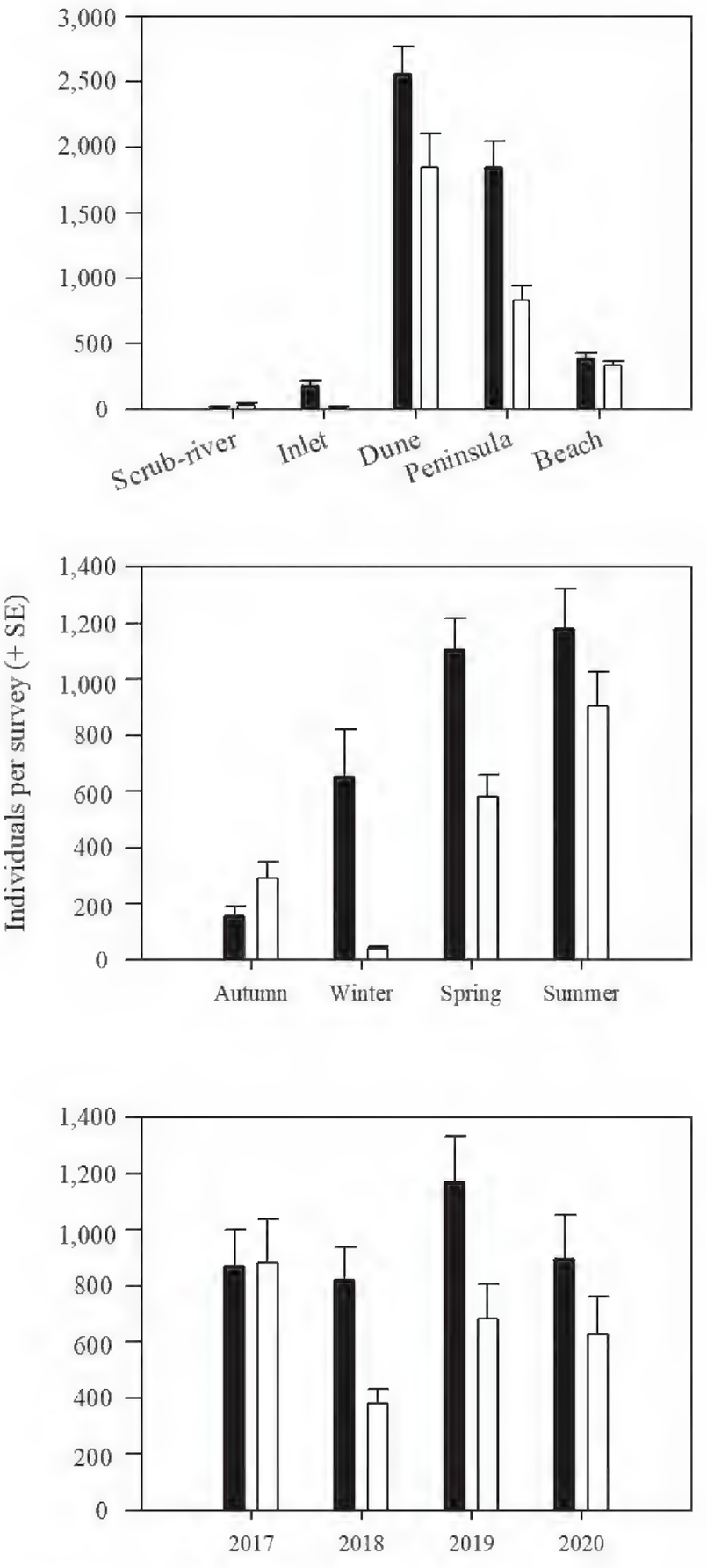

Figure 4. Laughing Gull (filled column) and Royal Tern (unfilled column) mean (+ SE) abundance among land cover types (top), seasons (middle), and years (bottom) surveyed at Huguenot Memorial Park, Florida (n = 129 surveys).

68.6 ± 2.0% and 68.0 ± 7.2% of the total numbers counted per survey, respectively. We recorded only Brown Pelican, Herring Gull, and Great Black-backed Gull juveniles in winter (Table 3). We observed nestlings, fledglings and/or juveniles of the American Oystercatcher, Black Skimmer, Herring Gull, Caspian Tern, Least Tern, Willet, and Wilson's Plover in <10 surveys per species, with greatest numbers recorded in the peninsula, beach, and inlet (Table 3).

The inlet was used by all listed birds and the Willet, Short-billed Dowitcher (*Limnodromus griseus*), Double-crested Cormorant (*Phalacrocorax carbo*), Dunlin (*Calidris alpine*), Great Egret (*Ardea alba*), Black-bellied Plover (*Pluvialis squatarola*), Semipalmated Plover (*Charadrius semipalmatus*), and Wilson's Plover (Tables 1 and 2). We recorded greater abundances per survey ($P < 0.030$) in spring and summer for the Willet and Great Egret, spring for the Wilson's Plover, autumn through spring for the Black-bellied Plover, autumn and winter for the Semipalmated Plover, winter for the Dunlin, and winter and spring for the Double-crested Cormorant.

Abundances per survey differed annually in the inlet and coastal scrub-riverside ($H_{inlet} = 7.91$, $H_{scrubriverside} = 29.80$, $P < 0.048$; Fig. 3). Among the land cover types surveyed, the coastal scrub-riverside had the fewest numbers of individuals (Table 1; Fig. 3). From the inlet surveys, Short-billed Dowitcher and Wilson's Plover were more abundant in 2017 than 2018 to 2020 ($P < 0.030$) and the Willet had fewer counts in 2018 than the other years surveyed ($P < 0.001$). In 2020 as compared to 2017, abundances were relatively low or not present for the Great Egret ($P = 0.0075$), Red Knot (*Calidris canutus*), Piping Plover, Tricolored Heron (*Egretta tricolor*), Little Blue Heron (*Egretta caerulea*), and Reddish Egret (*Egretta rufescens*). However, the Red Knot and Double-crested Cormorant had greater counts in 2019 as compared to 2018 and 2020 (Table 1) and the American Oystercatcher and Least Tern had greater numbers in 2020 (Table 2).

Discussion

The marsh area of Ft. George Inlet, the coastal scrub, banks of the St. Johns River, dunes, peninsula, and sandy beaches at Huguenot MP were used by the avian population for roosting, foraging, and nesting activities despite the impacts of flooding and high winds from extreme storm events. From 2017 to 2020, the inlet had the greatest species richness, which included all state or federally listed species, and the peninsula had greatest abundance. Each species used, on average, three of the five land cover types, which underscores the importance of managing natural resources to support resident, migratory, and breeding populations. The low-energy and intertidal mud and sand flats

located along the Ft. George Inlet, St. Johns River, and coastal beaches likely supported foraging and roosting needs (Lott et al. 2009). Gunnels (1999) reported that tidal height measured in intertidal sand and mud flats contributed up to 75% of variance of shorebird abundances. At Huguenot MP, the beaches were used by common (present in >50% of the 129 surveys) taxa such as the Herring Gull, Ruddy Turnstone (*Arenaria interpres*), and Sandwich Tern, and the peninsula's sandy flats and Ft. George Inlet's mud flats were used by Short-billed Dowitcher, Dunlin, Double-crested Cormorant, Black Skimmer, Black-bellied Plover, and Semipalmated Plover. By comparison, Black Skimmer and the Brown Pelican likely roosted in the park and foraged offshore (Lott et al. 2009). In addition, the Laughing Gull and Royal Tern were the most common and abundant species and largely responsible for greater abundances in the dunes and peninsula during nesting seasons.

Annual variability in the abundances of migratory and wintering birds was likely a function of local and regional changes in food and habitat availability and quality, human and predator interactions, and physical disturbance from storm events (Gunnels 1999, Sprandel 2007). Convertino et al. (2011) observed that physical disturbance from extreme storms may increase the availability of suitable habitats for species such as the Snowy Plover, Piping Plover, and Red Knot due to the redistribution of sand across dune complexes and removal of vegetation. Thus, hurricanes may have contributed to the recovery of the Herring Gull and Ring-billed Gull and the increased residence time of the Red Knot at Huguenot MP (February to May in 2017, May to December in 2018, and January to April in 2019). However, flooding from Hurricanes Irma (2017) and Michael (2018) may have contributed to lower avian richness in the coastal scrub-riverside and inlet areas. For example, counts of the Piping Plover, Wilson's Plover, Short-billed Dowitcher, and Tricolored Heron in 2020 had not returned to numbers recorded in 2017. To elucidate shifts in migratory and wintering bird populations recorded in Florida, local and regional ecosystem-level data should be collected in conjunction with avian diversity metrics (Jodice et al. 2019).

Of the 20 breeding shorebird and seabird species in Florida (FWC 2019), 13 were recorded in Huguenot MP during the study period. Although recorded year-round, the Laughing Gull, Brown Pelican, and Royal Tern had significantly greater counts during spring and summer when chicks were present, a seasonal pattern observed by Sprandel (2007) in the Florida Panhandle from 1994 to 2005. Brown Pelican chicks were first observed in the latter months of their migration season (Fink et al. 2020), suggesting that their breeding season may be extended at Huguenot MP. In contrast, an absence of Laughing Gulls in March and April, and fewer Brown Pelicans from May through August

were observed in east-central Florida (Stolen 1999). Despite extreme storm events, resources were available to support Laughing Gull and Royal Tern breeding populations at Huguenot MP.

In conclusion, the diversity of coastal land cover types in Huguenot MP supported foraging, nesting, and roosting activities of the avian population. Monitoring local and regional resource use by migratory and resident bird populations can be used to inform priority management actions. This study provides baseline patterns to evaluate future changes in the avian community as a function of predicted sea level rise, increased frequency and intensity of extreme storms events, and changes in invasive species, predator populations, and coastal development (Sprandel 2007, Convertino et al. 2011, Jodice et al. 2019, Vitale et al. 2020).

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ABUNDANCE AND PERSISTENCE OF SALTMARSH AND NELSON'S SPARROWS IN A REMNANT SMOOTH CORDGRASS TIDAL MARSH ON THE CENTRAL GULF COAST OF FLORIDA

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Abstract.—Saltmarshes dominated by smooth cordgrass (*Spartina alterniflora*) are currently rare in the Tampa Bay region and likely represent remnant patches of more extensive marshes that were extant during the nineteenth and twentieth centuries. We investigated small populations of two tidal marsh species, Saltmarsh (*Ammospiza caudacuta*) and Nelson's (*A. nelsoni*) sparrows, on one such isolated marsh (Shell Key) in Pinellas County, Florida. We sought to characterize basic population characteristics of relative abundance and persistence of individuals of the two species over a seven-year period. We color-banded sparrows to be able to identify all returning individuals, including those that we saw and were not captured on subsequent visits. In most cases, we visited the marsh twice (late autumn, late winter) to examine within-winter season persistence. We captured a total of 51 sharp-tailed sparrows (45% Saltmarsh and 55% Nelson's sparrows) from February 2010 to November 2016. We recaptured 11 individuals including three repeats in the same winter season and eight returns between years, thus providing wintering evidence (site tenacity) and evidence of philopatry, respectively, for both species. We estimated on average that four Saltmarsh Sparrows and six Nelson's Sparrows overwintered in the marsh based on mean annual numbers of the two species captured and observed bypassing nets during the February visit. A female Saltmarsh Sparrow banded during our study was recaptured at a nest with four eggs in the ensuing breeding season on a saltmarsh in Warren, Rhode Island. This study is the first to document more than 4% of Saltmarsh Sparrows surveyed on the central Gulf coast of Florida in recent decades. It provides an explanation for why current field observers have found so few Saltmarsh Sparrows in recent decades along this coast where historically they were regarded as common and widespread in the late 1880s in Tampa Bay area tidal marshes. Saltmarsh Sparrows evidently are narrower habitat specialists along the Gulf Coast than are Nelson's Sparrows. The former species locally prefers smooth cordgrass marshes, which have largely disappeared because of human development and black mangrove (*Avicennia germinans*) intrusion; the latter species uses the same marsh type and also large stands of black needlerush (*Juncus roemerianus*) that Saltmarsh Sparrows largely avoid. We hypothesize that Saltmarsh Sparrows perhaps established an old connection to tidal marshes dominated by *S. alterniflora* along the central Florida Gulf coast during the last Pleistocene ice age glaciation event when suitable tidal marshes were more widespread before mangroves spread northward into the region.

Key words: *Ammospiza caudacuta*, *Ammospiza nelsoni*, Florida, Nelson's Sparrow, non-breeding, remnant habitat, Saltmarsh Sparrow, smooth cordgrass, *Spartina alterniflora*, threatened extirpation

Tidal marshes provide valuable ecological services that benefit humans. They support fisheries, store carbon, protect our coastlines, and cleanse coastal waters (Viosca 1928, Zedler and Kercher 2005, Barbier et al. 2011). Tidal marshes also provide habitat for coastal marshland specialists, many of which are threatened by sea-level rise associated with climate change. In subtropical climates, black mangrove (*Avicennia germinans*) forests, which are currently advancing northwards, often out-compete the more tidally influenced, grassland-dominated saltmarshes when the absence of winter freezing enables mangroves to flourish (Duke et al. 1998). In the Tampa Bay area, saltmarshes dominated by smooth cordgrass (*Spartina alterniflora*) or by black needlerush (*Juncus roemerianus*) and mangrove forests all occur today within a similar geomorphic setting (Friess et al. 2012). This area is also the southern extent of the range of smooth cordgrass saltmarshes on the Florida Gulf Coast and the northern extent of wetlands dominated by mangroves during the late nineteenth century (Raabe et al. 2012). Mangroves expanded north along the Florida coasts during the Holocene and displaced most saltmarshes dominated by smooth cordgrass (Saintilan et al. 2014). As warmer winters continue to develop, mangrove forests are predicted to continue extending their range northward and likely will reduce saltmarshes even further in this area (Osland et al. 2013). Although populations of saltmarsh-dependent species may be able to survive in reduced numbers in diminishing patches of marsh grass, inevitably they will disappear locally and regionally as remnant patches are overrun by intruding mangroves. We believe we are now witnessing this scenario happen in the Tampa Bay area.

Nelson's Sparrows (*Ammospiza nelsoni*) in the Tampa Bay area appear to be historically derived from an inland continental area and populations apparently later spread into brackish marshes eastward to the Atlantic Coast (Greenlaw 1993, Shriver et al. 2020). Saltmarsh Sparrows (*A. caudacuta*) apparently originated along the north-central Atlantic Coast in tidal marshes (Greenlaw 1993, Greenlaw et al. 2020) and have become a classic saltmarsh specialist. The Saltmarsh Sparrow currently is believed to be under extreme threat from sea-level rise and its existence may become critical by mid-century (Hanson and Shriver 2006, Bayard and Elphick 2011, Field et al. 2017).

At this time, information on wintering ecology of these marshland sparrows, notably in Florida, is severely limited, even regarding basic population data on habitat selection, abundance, and persistence (Greenlaw et al. 2020, Shriver et al. 2020). No reports on local populations

exist for the two species on the Atlantic or northwestern Gulf coasts of Florida. One recent general investigation based on a critical specimen analysis examined wintering distributions and migration of subspecies of both species along the Atlantic and Gulf coasts including Florida (Greenlaw and Woolfenden 2007). Two local studies of tidal marsh communities of the sparrows on the peninsular Gulf Coast of Florida were published in recent years on their population ecology in Pasco and Levy counties (Post 1998, Tracey and Greenlaw 2009). These two studies, one in the Tampa Bay region and the other northward on the coast, provided information on basic population characteristics, habitat selection, subspecies composition, and breeding geographic origin of the sparrow communities.

Our interest in the Shell Key birds in the Tampa Bay area was elicited initially by the small size of the marsh patch used by the sparrows. Studies of Saltmarsh Sparrows in southern New England (Benoit and Askins 2002, Meiman et al. 2012) and Nelson's Sparrows in northern New England and the Maritimes (Hanson and Shriver 2006) document a positive relationship between relative breeding abundance and marsh size in these species. However, no study demonstrated an explicit lower limit in marsh size below which the breeding species were always absent. Nelson's Sparrows in northern coastal areas were either absent or in small numbers in marshes of about 1–2 ha, but response to marsh size may not be as important in the nonbreeding period when only survival is at risk.

Saltmarsh Sparrows were reported in the Tampa Bay area at the mouth of the Anclote River (Tarpon Springs) as early as the late nineteenth century. Scott (1889) regarded the species that we know today as Saltmarsh Sparrow (Greenlaw and Woolfenden 2007) as rather common in the Tarpon Springs area at that time. Greenlaw and Woolfenden (2007) examined specimens collected by W.E. Scott and others taken from this early period. Based on those specimens, Saltmarsh Sparrows were at first exclusively present in this historical sample on the Gulf Coast from Pinellas and Pasco counties in contrast to other locations along that coast. Specimens of Nelson's Sparrows did not appear until later in the early twentieth century (Tracey and Greenlaw 2009:table 2). In recent time a few Saltmarsh Sparrows in winter are encountered as a few individuals at one or two sites, otherwise at most sites they occur only rarely as solitary individuals that appear and disappear quickly (Smith 2011). Tracey and Greenlaw (2009) photographically documented 153 Nelson's Sparrows in wintering marshes dominated by black needlerush along the coast of Pasco County and only peripherally into the neighboring county to the north, but they documented only two Saltmarsh Sparrows in this marsh-type during the two years of their study (2006–2008).

We investigated the sparrow community on the very small marsh on Shell Key in Tampa Bay to document the abundance and persistence of Saltmarsh Sparrow relative to Nelson's Sparrow in such a small marsh patch. At the time we undertook this study on the Shell Key populations of sparrows (2010), we had no evidence that the Saltmarsh Sparrow still wintered in the Tampa Bay area. But Nelson's Sparrows were documented wintering nearby in the area (Tracey and Greenlaw 2009). This information in relation to the study by Tracey and Greenlaw (2009) raised the question whether Saltmarsh Sparrows were more habitat-specific than Nelson's Sparrows. Also, we were uncertain whether a marsh as small and isolated as Shell Key might harbor overwintering individuals of either species. Our chief objective was to examine the status of both species on the remnant marsh of Shell Key under the hypothesis that wintering populations of both species could sustain themselves on the local food resources there under survival-only risks. The persistence issue was important to us to ensure that the sparrows were not simply stopping at this small patch marsh temporarily during migration. This meant we had to sample the populations over a multi-year period and minimally twice during a typical wintering cycle to fully assess persistence—once in the fall after arrival and again in mid-winter before molt began, after which molting would likely hold them until spring departure. We needed to color-mark the two species to be able to identify them individually to ensure we also recognized birds that had been previously captured and had returned but not recaptured. Thus, we had to document persistence in terms of some individuals remaining on Shell Key during single winter cycles and also returning in subsequent years. Finally, we sought to estimate the wintering abundance of the two species on Shell Key and to determine the evident geographic sources of individuals of the two species for comparison with another earlier study in Levy County to the north of the Tampa Bay region (Post 1998).

Methods

Study site.—The study area (initially 0.32 ha; 27.6649 N, –82.7419 W) was located on Shell Key (~789 ha in size) south of Pass-a-Grille Beach and west of Tierra Verde, Pinellas County, on the west side of Tampa Bay (Fig. 1). The study area consisted of two cordgrass marsh patches (smooth cordgrass was dominant with scattered intruding mangrove seedlings), a northern larger patch (~50 m × 25 m) and a smaller one half the size separated by a small cove about 75 m wide, which the sparrows readily crossed when they were pushed by observers towards mist nets. On the landward (barrier island) side of the marsh, black mangroves formed a dense, narrow zone at the edge of a bay. The study began in January 2010 and ended in November 2016, on a visitation schedule of mostly twice each winter cycle in mid- to late autumn and mid-winter. So far as we knew, the patches of smooth cordgrass marsh comprising our study area were the only such patches in Shell Key Preserve (including islands around Shell Key) or in the immediate area. Most

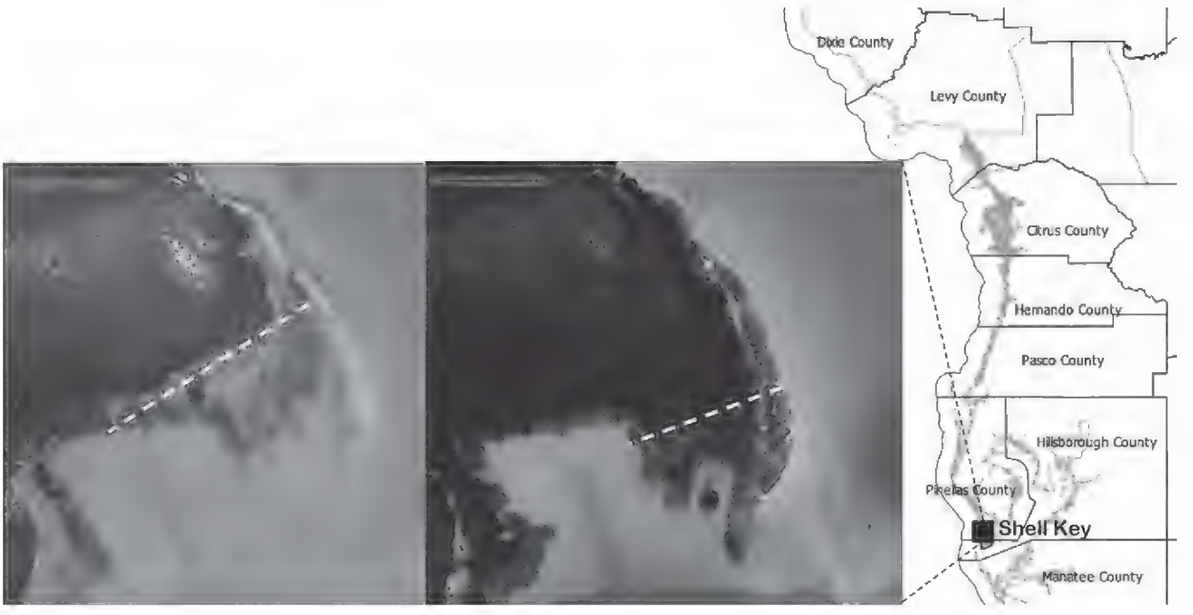

Figure 1. The central and northern peninsular Gulf Coast of Florida showing counties and the location on Shell Key in northwest Tampa Bay, Tampa, Florida. Two aerial photographs of Shell Key marsh (study area) show approximate boundary of black mangroves (dashed lines) (leftmost, April 2010, rightmost, January 2017).

of the vegetated inner intertidal zones and shorelines on Shell Key and all sections of small neighboring islands in the bay were occupied by mangroves.

Capture and banding methods.—Capture and banding activities took place during the lowest tides of the month to ensure that the maximum amount of saltmarsh was exposed (Table 1) and that some sparrows were not waiting in cover for water levels to subside under other tidal conditions. Under conditions of a low spring tide, sharp-tailed sparrows should be foraging in the saltmarsh grasses and on small, open patches in the marshes. We set two to four 12-m mist nets in a U formation along the downwind edge of the saltmarsh and the adjacent mangrove. Netting was mostly concentrated on the large marsh patch where the sparrows centered their activities. We also netted on the small patch if several sparrows retreated there as we flushed sparrows towards the nets on the larger one. Volunteers, including experienced banders, participated in flushing the sparrows towards the nets. We immediately removed any captured birds and placed a United States Fish and Wildlife Service aluminum band on each bird along with a unique color combination of plastic leg bands to facilitate individual identification of perched individuals during subsequent visits. We measured body characteristics including mass, wing chord, tail length, tarsus length, bill length from nares, and a fat score (0–7) for each bird. JSG identified subspecies during the banding process and later verified the determination from measurements and photographs. Subspecies for each sharp-tailed sparrow species followed diagnoses provided in Greenlaw and Woolfenden (2007). We completed one to three drives during each sampling event dependent on duration of low tide and wind conditions.

To measure persistence, for our purposes we employed two metrics: repeat recaptures (site tenacity) in the study area between seasonal visits (fall, mid-winter) in any one wintering period and returning recaptures between years (philopatry). We estimated the mean number (new + recaptures) of each species potentially wintering per visit as follows: given the total number Saltmarsh and Nelson's sparrows captured each winter season, we first calculated the proportion each species represented in the study area (11/27 = 0.41, 16/27 = 0.59, respectively). We then standardized the proportion of esti-

Table 1. The record of new captures and recaptures of Saltmarsh Sparrows and Nelson's Sparrows on a Shell Key tidal marsh from 28 February 2010 to 20 November 2016, Pinellas County Florida. Recaptures include repeat captures (autumn and mid-winter of the same nonbreeding season; site tenacity) and returns (between nonbreeding seasons; philopatry).

Season	Dates	Saltmarsh Sparrow	Nelson's Sparrow	New	Repeats	Returns[a]	Visits
2009–2010	28 Feb 2010	2	2	4	0	0	1
2010–2011	30 Oct, 6 Feb	4	3	7	1	0	2
2011–2012	30 Oct, 26 Feb	4	10	14	1	2 (2, 0)	2
2012–2013	4 Nov, 16 Feb	1	1	2	1	1 (1, 0)	2
2013–2014	15 Oct, 16 Feb	1	5	6	0	1 (0, 1)	2
2014–2015	26 Oct, 8 Feb	0	7	7	0	2 (0, 2)	2
2015–2016	31 Oct 2015	7	0	7	0	0	1
2016–2017	20 Nov 2016	4	0	4	0	2 (0, 2)	1
Totals		23	28	51	3	8 (3, 5)	13
Percent[b]		45%	55%				

[a]In parentheses we provide the number of Saltmarsh Sparrow returns, number of Nelson's Sparrow returns, respectively.
[b]Percent total original captures.

mated net-avoiders (1–5, mean of 3/visit) per visit by species (1.2, 1.8, respectively) and added the respective values to the total number of captured individuals per visit of each species. Finally, we rounded up the final mean value per visit to a whole number of estimated individuals of each species wintering during an average visit.

We used a chi-square contingency test with Yates correction to determine whether the proportions of the two species captured differed between fall or winter (within season). We also employed a linear regression analysis to test the hypothesis that sharp-tailed sparrows collectively exhibited no annual trend in total fall or winter numbers captured during seven years of fall visits and the six years of winter visits.

We were not working under the authorization of any institution. We adhered to the ethical standards of traditional, long-established banding authority rules (Bird Banding Laboratory) to carefully handle birds in the field, measure them, then immediately release them unharmed. The people who removed the birds from nets and handled the birds during banding and measurement were experienced banders. We lost no birds during capturing and handling (U.S. Geological Survey permit: 23458, Florida Fish and Wildlife Conservation Commission permit: LSSC-10-00069A-C, LSSC-14-00034A-B).

Results

During a total of 13 mid- to late fall and mid-winter visits to Shell Key from 28 February 2010 to 20 November 2016, we captured and banded a total of 51 sharp-tailed sparrows. Of this total, we recaptured eight returning individuals previously banded at the site. Of the total individuals captured, 55% were Nelson's Sparrows and 45% were Saltmarsh Sparrows (Table 1). Considering only sparrows that were known to have returned (we may have missed others that we did not

detect), the minimum observed return rate of Nelson's Sparrows was about 11% (3/28) and of Saltmarsh Sparrows was 13% (3/23). In addition, one Saltmarsh Sparrow originally banded at Shell Key (2511-39150) on 31 October 2015 was encountered on its breeding grounds by northern observers on 16 June 2016, when it was photographed as a female with a nest of four eggs in the tidal marsh at Jacobs Point Preserve, Warren, Rhode Island. The female was subsequently captured and the band number was confirmed.

We found evidence of both site tenacity and philopatry in both species. Site tenacity (same season repeats) involved two cases in Saltmarsh Sparrow and only one case in Nelson's Sparrow. Five cases of philopatry (returns in different seasons) occurred in Nelson's Sparrow and three cases in Saltmarsh Sparrow (Table 1). Two Nelson's Sparrows returned twice during the study. Based on mid-winter capture numbers and a conservative estimate of the number of individuals that escaped capture, we estimated that on average four Saltmarsh Sparrows and six Nelson's Sparrows overwintered on Shell Key in any one year. Minimum and maximum numbers of both species present on any one visit varied between visits and years from about five to about 15 (during the fall, reflecting migrant influx).

Total numbers of sparrows of each species that we captured in mid-winter (including recaptures, n = 11 Saltmarsh Sparrows, 16 Nelson's Sparrows) was similar to the numbers captured in the autumn (n = 18, 17, respectively; chi-square with Yates correction: $\chi^2 = 0.336$, df = 1, $P = 0.56$) in spite of apparent migrant arrivals in the fall; small sample size may have been a factor in this analysis. Mean number of Saltmarsh and Nelson's Sparrows wintering per visit was 3 and 4, respectively, varying across both species from 2 to 5. Because we failed to find or capture any Saltmarsh Sparrows on the marsh during our two earliest visits in October (15, 26 October), we suspect that the typical schedule of arrival of the two species differed on average at this location, although arrivals of each species may have varied annually from mid- to late October. Yet 31 October appeared to represent a recent surge of arrivals of one or both species in two visits on that date (Table 1). We detected no trend of sharp-tailed sparrows (both species combined) during the study period (2010–2016) within either fall season (linear regression, slope ± SE: 0 ± 0.870; $P > 0.99$) or late winter season (0.2 ± 0.263; $P = 0.49$).

JSG examined each banded sharp-tailed sparrow in the field and made subspecies determinations for 49 of 51 (2 unknowns) sharp-tailed sparrows. All Nelson's Sparrows represented the nominate, interior population from the northern Great Plains, *A. n. nelsoni*, and all except one of the Saltmarsh Sparrows exhibited plumage characters typical of the nominate subspecies, which occurs in the northern portion of

the species' breeding range. The single exception was a distinctly dark-plumaged individual that may have been *A. c. diversa* from the southern portion of the breeding range. Bill length from the anterior margin of the external nares for respective subspecies between species (longer in Saltmarsh Sparrows than in Nelson's Sparrows) supported plumage characteristics in each case. One individual Saltmarsh Sparrow examined on 30 October 2011 showed evidence of wing (inner greater secondary coverts) and tail molt. All other individuals of both species were in fresh body plumage with no evidence of molt.

Discussion

Apart from Post's (1998) study, little previously was known on the current status and distribution of sharp-tailed sparrows along the western coast of peninsular Florida. Post's (1998) investigation of a large area of marshland north of the mangrove zone in the Tampa Bay region found that both of these species formed loose flocks in winter in the marsh patch type dominated by smooth cordgrass, which covered 38% of the study area. There was no mention of either species also using or avoiding pure stands of black needlerush, which occupied 26% of the study site. Generally the latter plant grows in upper reaches of tidal marshes where both plants occur in the southeastern United States (Touchette et al. 2009). Saltmarsh Sparrows comprised only 4% of the wintering sharp-tailed sparrow community on Gulf Hammock in Levy County; thus, they were relatively uncommon at that latitude (29.175° N) as they are north to Wakulla and Franklin counties (Post 1998). In contrast, on a remnant marsh in Tampa Bay, we found Saltmarsh Sparrows to be relatively more abundant than Post (1998) found farther north on well-developed marshland also dominated by smooth cordgrass. The minimum average return rate of Nelson's Sparrows on Gulf Hammock was 15%. The return rates of each of the two species on Shell Key, considering the small sample sizes, were similar to those reported by Post (1998) for Nelson's Sparrows. We captured one Nelson's Sparrow returnee in both its first and third years, whereas all other recaptures involving both species returned during the season following the banding season. Post (1998) speculated that the low return rate of Nelson's Sparrows on Gulf Hammock was likely due to low annual survival of first-year birds. He found only 6% after-hatch-year birds in a sample of autumn captured individuals. We did not assess the ages of birds that we captured in October, but our low return percentages perhaps also reflected chiefly young individuals in their first year at Shell Key.

An examination of the distribution of nonbreeding sharp-tailed sparrows, based on mostly historical specimens collected in the late

nineteenth and early twentieth centuries (Greenlaw and Woolfenden 2007), found that Saltmarsh Sparrows occurred in autumn and winter along the northwestern coast of peninsular Florida from Pinellas and Pasco counties in the Tampa Bay region north to Wakulla County at the eastern end of the panhandle. A notable historical pattern of occurrence found by Greenlaw and Woolfenden (2007) on this coast was that Saltmarsh Sparrow in Scott's series of specimens that we examined was the only species taken during the years that he collected in the Tarpon Springs area. Scott (1889) reported that the "Sharp-tailed Finch" (an old name [Samuels 1870], as Scott listed the species) was "a rather common" migrant and winter resident on the marshes in his area. He initially sent six specimens to J. A. Allen for his diagnosis (Allen 1888), and later sent 11 others (Scott 1889). Allen responded (in litt. to Scott) in relation to Scott's overall "Tarpon Springs" sample by stating that the Florida specimens differed only slightly from Long Island (New York) and New Jersey birds including some "easily" matching his local specimens, and that they warranted no separation from *Ammodramus caudacutus*, as the Saltmarsh Sparrow then was known. Greenlaw and Woolfenden (2007) later examined 15 of Scott's specimens and independently confirmed that all were Saltmarsh Sparrows. In contrast, Post's (1998) study farther north found both species of sharp-tailed sparrows associating in Gulf Coast marshes on the peninsular coast. But one must keep in mind that Nelson's Sparrow was already known (Allen 1875) at the time Scott was collecting and was treated in the American Ornithologists' Union Check-list (AOU 1886) as a distinctive subspecies of the species now recognized as Saltmarsh Sparrow. Thus, the report that Allen (who was the authority on Nelson's Sparrow of that early era), sent to Scott was significant in that he affirmed no Nelson's Sparrows were in Scott's series, as Greenlaw and Woolfenden (2007) also discovered. Indeed, a breakdown of determinations of specimens taken in Pinellas and Pasco counties from Scott's time to nearly mid-twentieth century, summarized from raw data of Greenlaw and Woolfenden (2007) in Tracey and Greenlaw (2009:table 2), suggest the first Nelson's Sparrows recorded from the Tampa Bay area were eight specimens from Pinellas County obtained by William G. Fargo beginning in 1925. Before then, Greenlaw and Woolfenden (2007) had examined a total of 20 specimens (from four different collections) obtained between 1888 and 1913, all of which were Saltmarsh Sparrows. Fargo's series of 34 specimens from the 1925 to 1934 period recorded 24% Nelson's and 76% Saltmarsh sparrows, which differed substantially from Post's (1998) more northerly sample decades later. These data suggest that the status of Nelson's Sparrows on the central Gulf Coast of Florida may have shifted from none or rare in the late nineteenth century to uncommon during the early twentieth

century to even more common some decades later (Post 1998, Tracey and Greenlaw 2009, this study). In addition, also in the Tampa Bay region in Honeymoon Island State Park, Pinellas County, both species were recently found in small numbers (one Saltmarsh Sparrow in each of two years and up to six Nelson's Sparrows during early winter from 2008–2016) during the Northern Pinellas Christmas Bird Count in a limited section of narrow (≤0.5 m), fringing strips of smooth cordgrass along the bayside of the spit of land in the park.

Thus, our study of a small, remnant and isolated saltmarsh in southern Pinellas County is the first in recent time to document more than 4% of Saltmarsh Sparrows in surveys of populations of sharp-tailed sparrows on Florida's central and northern peninsular Gulf Coast. Clearly and notably, this study confirms that a relatively high proportionate number of Saltmarsh Sparrows, albeit in a very local population, still visited a patch marsh that is disappearing in the study period as black mangroves intrude. Given the general absence of stopover tidal marshes in the southern half of the Florida peninsula on both coasts, and given the dominance of black needlerush in those marshes that persist in estuaries on the southern Gulf Coast, we hypothesize that migrant Saltmarsh Sparrows reach the west side of the peninsula by making nonstop, overland flights from the northeast Atlantic Coast. Nano-tag banding of sparrows on their breeding grounds can test this hypothesis. We conclude that Saltmarsh Sparrows are largely restricted to smooth cordgrass-dominated tidal marshes on the west coast of Florida unlike Nelson's Sparrows, which exhibit a broader habitat association in both smooth cordgrass and black needlerush marsh patch types. This conclusion seems clear because the species rarely occurs in black needlerush stands in this region—the only other tidal marsh type in the region (Tracey and Greenlaw 2009)—and because medium height smooth cordgrass is often used by the species for nesting in the upper intertidal on its breeding grounds (Greenlaw et al. 2020) and in winter along the Atlantic Coast north of Florida (Watts and Smith 2015; W. Post in litt. to JSG, South Carolina).

Although the Shell Key sample sizes were small, they appear to indicate a higher-than-expected current proportion of Saltmarsh Sparrows on the central-west coast of Florida, perhaps still mirroring the historical pattern and contrasting with proportionately fewer numbers farther north towards the limits of the species' Florida wintering range on the Gulf Coast. The observations of Saltmarsh Sparrows on Shell Key and on even smaller fringing, bayside strips of smooth cordgrass on Honeymoon Island suggest that minimum marsh sizes apparently influencing populations in the breeding season may not apply to non-breeding sites on the west coast of Florida and perhaps not elsewhere in winter. In southern New England, breeding

Saltmarsh Sparrows were less abundant or absent in small marshes (<10 ha) than in more expansive ones (Benoit and Askins 2002).

The apparent prevalence of Saltmarsh Sparrows on the central Gulf Coast of Florida historically (Greenlaw and Woolfenden 2007) suggests that smooth cordgrass marshes were larger or more prevalent during the late nineteenth and early twentieth centuries than today. As noted, W. E. D. Scott's and W. Fargo's collections of series of Saltmarsh Sparrows suggest significant areas of tidal marsh dominated by smooth cordgrass existed there during that period, while they have been developed out of existence since then or have been overtaken by mangrove intrusion. Other independent evidence also indicates that grassland tidal marsh dominated by smooth cordgrass in the outer salinity zone was more prevalent in the Tampa Bay region in the nineteenth century than in the twentieth century (Raabe et al. 2012). Today, black mangroves are becoming prevalent well north of Tampa Bay. In the bay these mangroves appear to have erased other patches of such marsh in the area and at Shell Key this process is nearing complete marsh overrun. In 2010 at the start of our study, we estimated the size of the Shell Key marsh to be 0.32 ha (Global Earth tool, Google Earth; Google, Mountain View, CA). By 2015, we estimated the size to be 0.16 ha, but sparrows were still using what remained in winter.

We suggest that the long period of low sea levels during the last Pleistocene ice-age glaciation cycle and ensuing Holocene sea-level rise across the broad, formerly exposed Gulf-side continental platform in Florida may have been a time of extensive tidal marsh development before subtropical mangrove intrusion became an issue. Although extensive cordgrass tidal marsh may not have been required for some individuals of north Atlantic coastal populations of Saltmarsh Sparrows to cross land in Florida to establish a migratory route to the Gulf of Mexico, larger areas than now exist must have been necessary for this to happen. We suspect that prevalent, high-quality cordgrass tidal marsh on the central Gulf Coast in recent Holocene time would be sufficient to establish a population level. But we also suggest that a Gulf Coast route is likely an older pattern. The period of low sea levels and cool climate during the late Pleistocene may have been a time when cordgrass marshes probably were prevalent in Florida when Saltmarsh Sparrows could have consolidated a historical autumnal migration connection with the central Gulf Coast (and perhaps southward) from the Atlantic Coast. In any event, what may be an old historical wintering population of Saltmarsh Sparrows in the central Gulf Coast region of Florida now appears destined to extirpation, if it has not already happened. For a time, individual occurrences resembling vagrancy may be the last vestige of this old pattern in the Tampa Bay region.

Acknowledgments

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COASTAL OBSERVATIONS OF ALLIGATOR SNAPPING TURTLES IN THE FLORIDA PANHANDLE

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The genus *Macrochelys* (alligator snapping turtles) inhabits river systems that empty into the Gulf of Mexico from Florida to Texas and contains the largest freshwater turtles in North America (Pritchard 2006). This paper details observations of *Macrochelys* on the coast and barrier islands in Franklin, Gulf, and Bay counties, Florida, that contribute to our knowledge of their movements in brackish or saltwater ecosystems and occurrence on islands. Thomas et al. (2014) described turtles from this geographical area as a new species, Apalachicola alligator snapping turtle (*M. apalachicolae*), but Folt and Guyer (2015) disputed this designation, and many authorities still consider them to be alligator snapping turtles (*M. temminckii*; e.g., Crother 2017, Turtle Taxonomy Working Group 2017).

We report five unpublished *Macrochelys* records from St. Vincent Island, the first four records from Dog Island, the first record from Little St. George Island, three records on or near Gulf County beaches, and one record from a Bay County beach (Table 1). From east to west in Franklin County, the following islands form a barrier-island chain in the Gulf of Mexico that partly encloses St. George Sound, Apalachicola Bay, and St. Vincent Sound: Dog, St. George, Little St. George, and St. Vincent (Fig. 1). The Apalachicola River provides approximately 90% of freshwater input to Apalachicola Bay, and fresh water is primarily exchanged with gulf salt water at four inlets: East Pass, Sikes Cut, West Pass, and Indian Pass (Huang and Spaulding 2002; Fig. 1). River discharge and exchange through these inlets primarily determine the flushing rate and salinity variations in the bay (Dulaiova and Burnett 2008). Model simulations indicate that the residence time of fresh water entering the bay from the Apalachicola River typically ranges between 3 and 9 days (Huang and Spaulding 2002).

The westernmost island in the chain of four barrier islands is 5,056-ha St. Vincent Island, which is located 10.5 km from the mouth of the Apalachicola River (Fig. 1). The island is part of St. Vincent National Wildlife Refuge and is accessible only by watercraft. St. Vincent Island has a maximum elevation of 2.1 m above mean sea level, and its primary landcover types are coastal grassland, mesic flatwoods, scrub, and tidal marsh.

Table 1. Observations of *Macrochelys* (from east to west) on the coast or barrier islands in Franklin, Gulf, and Bay counties, Florida. For turtles found on beaches, we provide the minimum distance by water from the nearest river mouth.

Location	Date	Status	Age class and sex	Nearest river mouth (distance)
Dog Island	18 Feb 2019	Live	Hatchling	Carrabelle River (12 km)
Dog Island	7 Jun 2019	Live	Juvenile or small adult	Inland
Dog Island	31 Dec 2018	Live	Adult	Carrabelle River (6.4 km)
Dog Island	7 Nov 2018	Live	Juvenile or small adult	Carrabelle River (8.2 km)
Little St. George Island	14 Sep 2018	Dead	Adult	Apalachicola River (15 km)
West Pass Beach[a]	8 Apr 1997	Dead	Adult male	Apalachicola River (17.5 km)
St. Vincent Island	18 Jan 2005	Dead	Adult	Inland
St. Vincent Island	Oct 2005	Live	Small juvenile	Inland
St. Vincent Island[a]	15 Aug 2000	Dead	Adult male	Inland
St. Vincent Island	25 Apr 2019	Live	Adult female	Inland
St. Vincent Island	25 Apr 2013	Live	Adult female?	Inland
Indian Pass	7 Nov 2011	Dead	Adult male	Apalachicola River (24 km)
Indian Peninsula	21 Feb 2021	Live	Juvenile	Apalachicola River (25 km)
S of Port St. Joe	1 Jun 2021	Live	Juvenile	Inland
WindMark Beach	Mar 2016	Live	Adult	Canal (2.2 km)
Mexico Beach	27 June 2021	Live	Adult	Canal (0.5 km)

[a]Lewis and Irwin (2001).

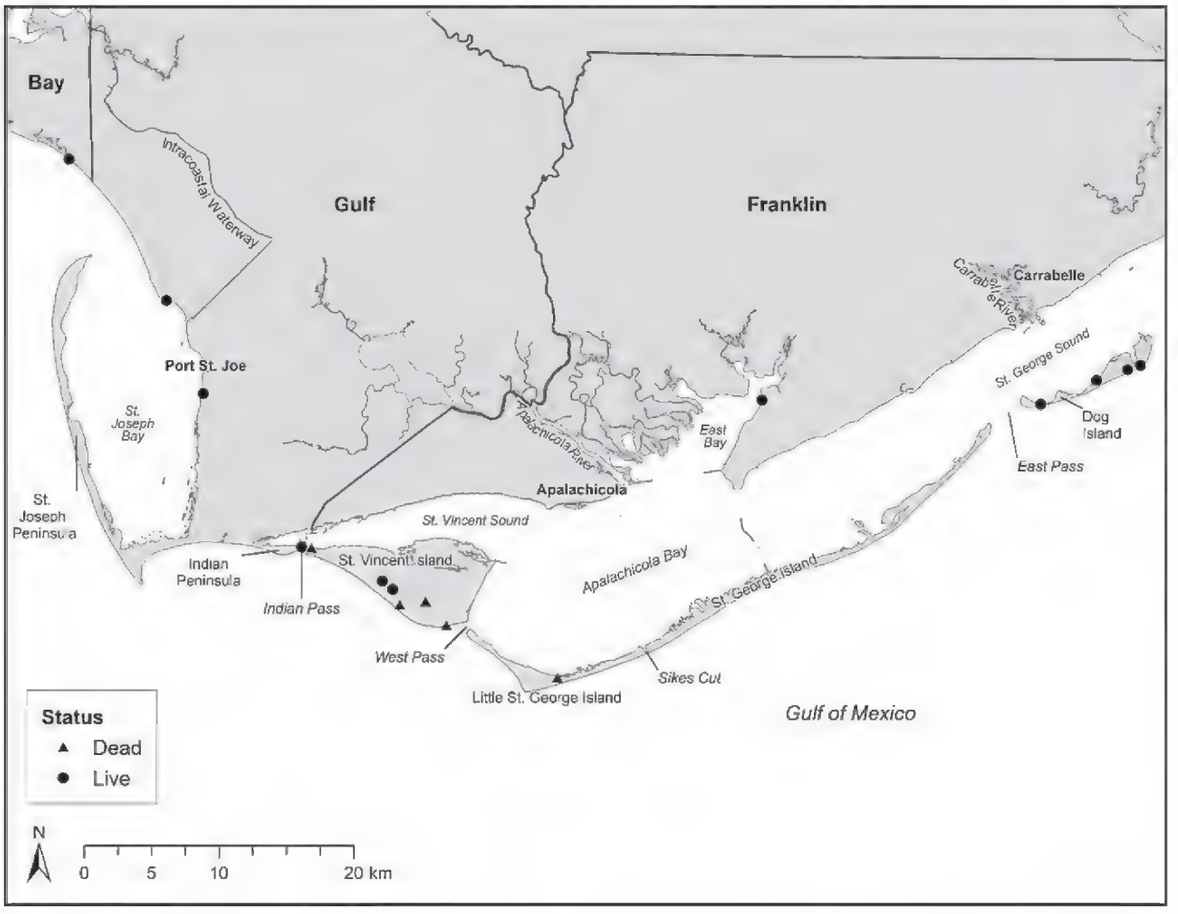

Figure 1. Locations of coastal and insular records of *Macrochelys* mentioned in this paper in Franklin, Gulf, and Bay counties, Florida.

It has a well-developed beach dune and swale system, sloughs (ephemeral, freshwater marshes in interdunal swales), and six permanent, freshwater coastal dune lakes. Lewis and Irwin (2001) reported two *Macrochelys* observations from the island, and we report five additional observations (Table 1; Fig. 1). Photographs were accessioned in the Herpetology Collection of the Florida Museum of Natural History (UF Herp; Gainesville, FL, USA). In August 2000, a dead adult male *Macrochelys* (specimen MHP 6811; Museum of the High Plains, Fort Hays State University, Sternberg Museum of Natural History, Hays, KS, USA) was found on the margin of dried-up Rattlesnake Slough, which extends westward from Oyster Pond, a coastal dune lake (Lewis and Irwin 2001). Thomas Lewis reported a dead adult turtle from Lake 5 (29.64638°N, 85.12876°W) on 18 January 2005 and a small juvenile (unknown location) in October 2005 (Florida Natural Areas Inventory database). On 25 April 2013, BSS photographed a probable adult female (UF Herp 191823; Fig. 2A) on B Road (29.65968°N, 85.16188°W). Her carapace bore extensive damage from American alligators (*Alligator mississippiensis*), which are abundant on the island. Given the date, she may have been on land to nest (Ewert and Jackson 1994). On 25 April 2019 at 0929 EDT, BSS photographed a nesting female (UF Herp 191822; Fig. 2B) along B Road (29.65408°N, 85.15396°W), confirming the presence of a reproducing population. The latter two turtles were approximately 2.2 km and 1.25 km, respectively, from the nearest coastal dune lake, Oyster Pond, but only 90 m from the same slough. Observations of all size classes and a nesting female on St. Vincent Island indicate an established *Macrochelys* population that uses coastal dune lakes and sloughs. According to the refuge's fishing regulations brochure, there are 110 ha of open fresh water, and four of the six permanent lakes have abundant largemouth bass (*Micropterus salmoides*)

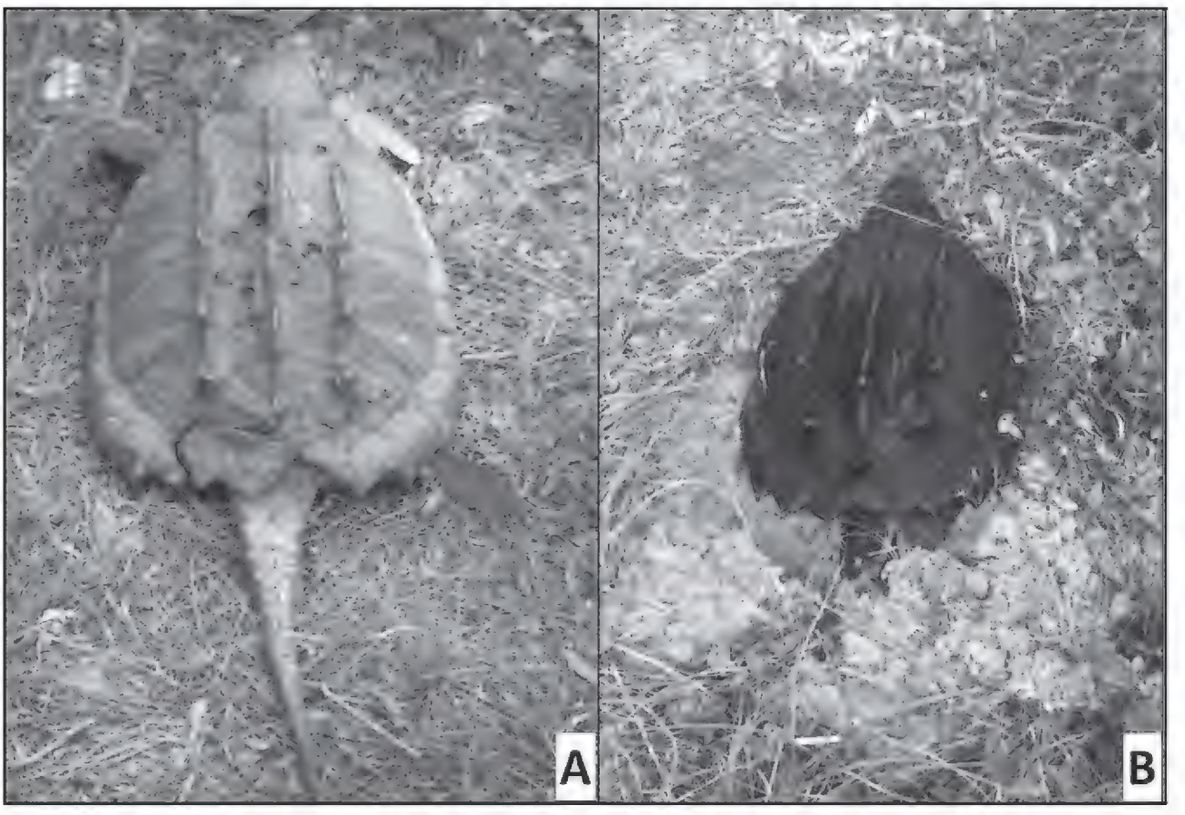

Figure 2. *Macrochelys* found on St. Vincent Island, Franklin County, Florida: A) probable adult female on B Road on 25 April 2013; B) female nesting along B Road on 25 April 2019 (photos by Bradley Smith).

and bluegill (*Lepomis macrochirus*) populations. Many of the wetlands are linked hydrologically, allowing turtles to move easily among them, especially during high water conditions. The refuge's water impoundment system also connects the larger wetland complexes. Lewis and Irwin (2001) speculated that the skeleton of an adult male found on West Pass Beach on St. Vincent Island in 1997 was transported there by currents and not part of an island population. This is probably also the case for a dead adult male (UF Herp 191824) found by BSS on 7 November 2011 on the beach near the bayside entrance of Indian Pass on St. Vincent Island (29.68120°N, 85.21660°W).

The next island to the east is the narrow, 14.4-km-long Little St. George Island, which was separated from St. George Island in 1954 when Sikes Cut was dredged for shipping. On 14 September 2018, DC and Christopher Matechik found an adult *Macrochelys* skull (29.59626°N, 85.02679°W) along the shore on the bay side of this uninhabited island (Table 1; Fig. 1). Florida Department of Environmental Protection staff have never had reports of stranded *Macrochelys* from St. George Island or Little St. George Island.

The easternmost island in this chain is 738-ha Dog Island, which is located 5.5 km offshore from the town of Carrabelle and at least 40.5 km from St. Vincent Island (Fig. 1). This island, which is 12.7 km long and up to 1.3 km wide, is mostly owned by The Nature Conservancy (Jeff Lewis Wilderness Preserve) and accessible only by watercraft or airplane. It has a similar elevation, beach dune and swale system, and assemblage of landcover types as St. Vincent Island, but its coastal dune lakes are shallower and much smaller (each <1 ha). Between November 2018 and June 2019, TC photographed four *Macrochelys* on Dog Island (Table 1; Fig. 1). On 7 November 2018, a juvenile or small adult (UF Herp 186721; Fig. 3A) was found partially buried on a barren, sandy spit on the gulf side at the western end of the island (29.78088°N, 84.65717°W). The turtle was released 7 km away in Firehouse Pond, a 0.6-ha coastal

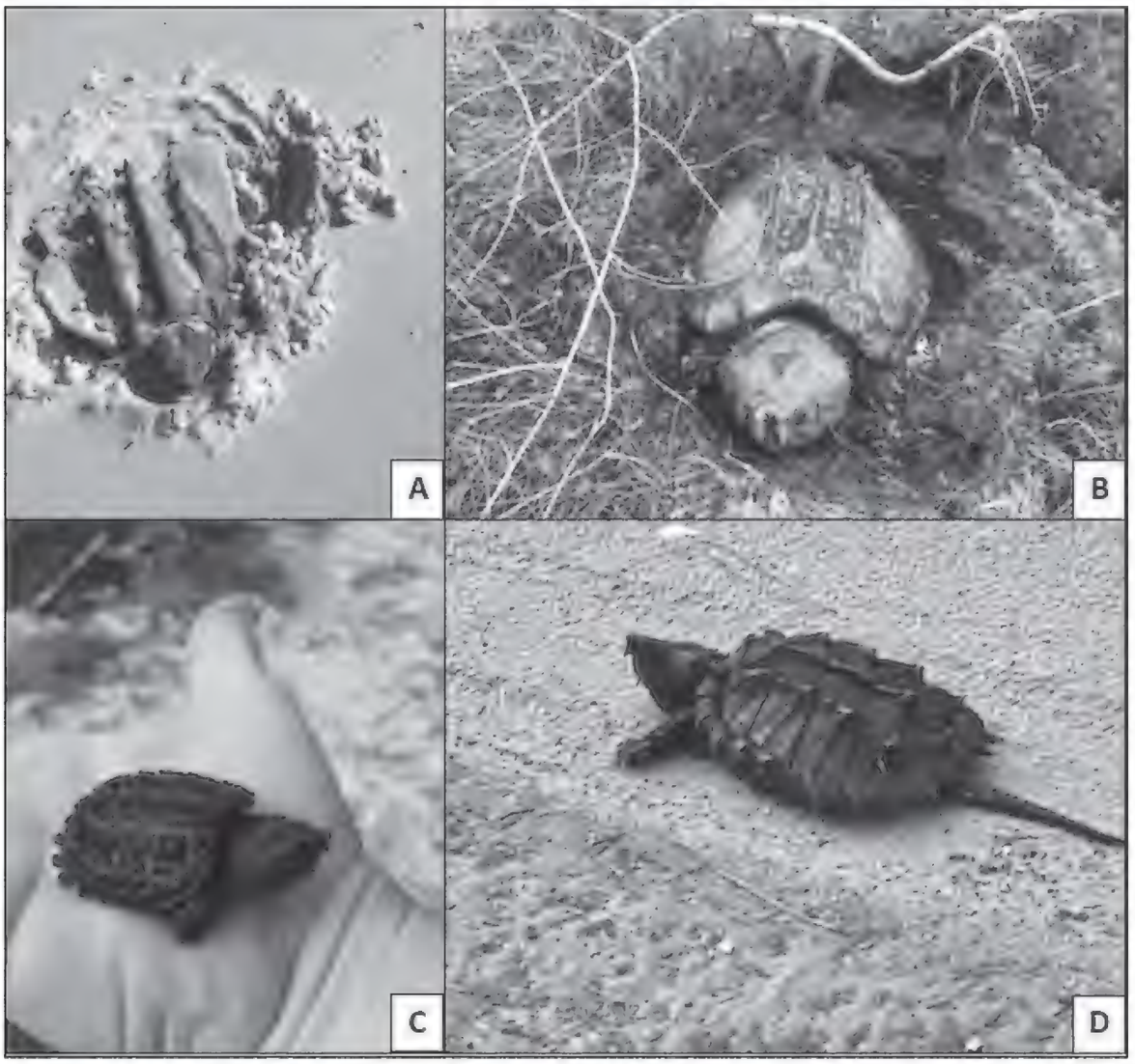

Figure 3. *Macrochelys* found on Dog Island, Franklin County, Florida: A) juvenile or small adult on the gulf side on 7 November 2018; B) adult on the bay side on 31 December 2018; C) hatchling from the surf on the gulf side on 18 February 2019, and D) juvenile on a road on the gulf side on 7 June 2019 (photos by Terri Cannon).

dune lake on the gulf side. On 31 December 2018, an adult with a deformed beak (UF Herp 187059; Fig. 3B) was found partially buried in sand in a resident's yard on the bay side (29.79702°N, 84.61448°W). Residents periodically poured fresh water on the turtle for 4 days until it was released 2.5 km away in Bream Pond, a 0.3-ha ephemeral coastal dune lake on the gulf side. On 18 February 2019, a hatchling (UF Herp 187908; Fig. 3C) was found in the surf on the gulf side (29.80720°N, 84.58066°W), about 60 m over a low dune from Bream Pond, where it was subsequently released after being held in captivity by Florida Fish and Wildlife Conservation Commission (FWC) staff for 128 days. This was the only *Macrochelys* collected by residents and provided to FWC for evaluation. On 7 June 2019, TC observed a juvenile or small adult (UF Herp 188729; Fig. 3D) crawling on a sand road about 50 m from Firehouse Pond (29.80412°N, 84.59057°W). This might have been the same turtle found on the beach 7 months earlier and released in Firehouse Pond. Another *Macrochelys* was seen about 15 years ago on Dog Island (TC, pers. obs.). Reported sightings of *Macrochelys* on Dog Island span 7.9 km (Fig. 1).

The relatively small and shallow coastal dune lakes on Dog Island completely or partially dry occasionally and would appear to be less suitable for *Macrochelys* than those on St. Vincent Island. Lakes on Dog Island have sand substrates with a shallow layer of organic deposits. The only cover consists of rushes (*Juncus* sp.), sedges (*Carex* sp.), cattails (*Typha* sp.), and sawgrass (*Cladium jamaicense*), which are primarily confined to the margins. These are oligotrophic systems with salinity levels that vary greatly depending on local rainfall and storms. Ephemeral Bream Pond is shallower and closer to the Gulf of Mexico than Firehouse Pond. We tried to determine whether a population was established on the island by setting six baited hoop net traps (122-cm diameter) in these two lakes for one night on 26 June 2019. We failed to catch anything and observed only small, minnow-like fishes. We had difficulty finding water sufficiently deep to submerge the trap funnel openings. Bream Pond was slightly brackish, whereas Firehouse Pond contained fresh water. The best evidence for an established population on Dog Island is the hatchling found on a gulf side beach across the dune from Bream Pond. If not hatched on Dog Island, this turtle would have needed to travel a minimum of 12 km through sea water from the mouth of the Carrabelle River to the location where it was discovered (Table 1; Fig. 1). Survival from predators and body mass loss associated with osmoregulation in saline water seems unlikely. Small turtles experience more rapid efflux of water via osmoregulation than large turtles (Dunson and Seidel 1986).

Additional observations of *Macrochelys* exist from beaches in the area (Table 1; Fig. 1). On 21 February 2021, Susan Vaughn found a juvenile *Macrochelys* (80-mm median carapace length; UF Herp 191774) along the high-tide wrack line of a beach on the gulf side of Indian Peninsula, Gulf County (29.68161°N, 85.22440°W), approximately 265 m across Indian Pass from St. Vincent Island. The presence of live water hyacinth (*Eichhornia crassipes*), a freshwater aquatic plant, washed up on the beach suggests the turtle may have rafted on floating mats flushed from the Apalachicola River during a high-water event. To reach the point where it was found near Indian Pass, the turtle would have had to travel a minimum of 25 km through Apalachicola Bay and St. Vincent Sound from the mouth of the Apalachicola River (Table 1). Because most of the fresh water discharged by the Apalachicola River exits to the gulf through West Pass, much of the journey presumably undertaken by this turtle would have been through low-salinity water, but salinity is high in St. Vincent Sound, particularly near Indian Pass (Livingston et al. 2000). On 1 June 2021, DC found a juvenile *Macrochelys* (~280-mm median carapace length; UF Herp 192142) crossing U.S. Highway 98, 215 m north of its junction with County Road 30A (29.78370°N, 85.30089°W) and south of the city of Port St. Joe, Gulf County (Table 1; Fig. 1). The turtle, whose carapace was scraped from encountering a vehicle, was 135 m from the shore of St. Joseph Bay and crawling east (inland). There was a small stream 50 m to the north that entered the bay from a swamp, but the turtle was traveling through a pine plantation. In March 2016, an adult *Macrochelys* was photographed in the surf on WindMark Beach (29.8455°N, 85.3300°W) north of Port St. Joe, Gulf County (Fig. 4). The turtle was on the mainland side of St. Joseph Bay about 200 m from a small, brackish tributary and 2.2 km from an 8.8-km-long canal that connects to the Intracoastal Waterway (Table 1). On 27 June 2021, Christy Adams photographed an adult *Macrochelys* (UF Herp 192242) in the surf in Mexico Beach, Bay County (29.9390°N, 85.4062°W), 500 m from the outlet of a small canal and 7 km north of the entrance of St. Joseph Bay (Table 1; Fig. 1). This represents the only coastal mainland record that was not in a bay.

The records above were from islands and on or near beaches. Although *Macrochelys* is reported to occasionally be found in brackish water, including Ochlockonee Bay, Franklin County (Ewert et al. 2006, Pritchard 2006), records in estuaries are difficult to document. A nest near the mouth of the Apalachicola River was adjacent to water sufficiently brackish to contain fiddler crabs (*Uca* sp.; Ewert and Jackson 1994). An alligator snapping turtle found near Mobile Bay, Alabama, supported barnacles thought

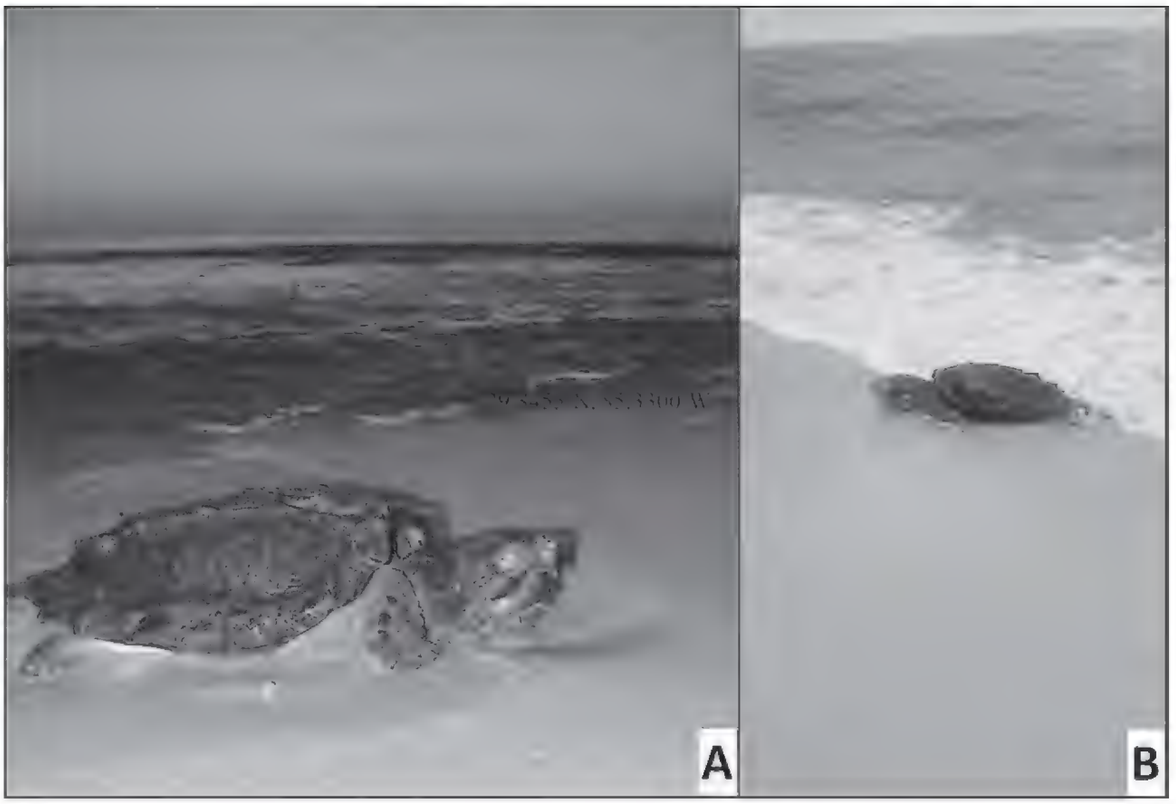

Figure 4. Adult *Macrochelys* in the surf A) in March 2016 on WindMark Beach, Gulf County, Florida (anonymous photographer), and B) on 27 June 2021 on Mexico Beach, Bay County, Florida (photo by Christy Adams).

to be several weeks old (Jackson and Ross 1971). Staff with FWC's Fisheries-Independent Monitoring (FIM) captured a *Macrochelys* 75 m offshore at the northern end of East Bay while sampling with a 183-m-long seine net on 20 March 2006 (Fig. 1). Several streams empty into East Bay, where the salinity is approximately 10 ppt (Sun and Koch 2001). Staff from FIM captured three other *Macrochelys* (not shown in Fig. 1) with a 6.1-m otter trawl in January 2009, April 2012, and May 2019 in tidally influenced streams within 500 m of East Bay (FIM inshore database).

Transport by humans is unlikely to be the source of *Macrochelys* on these islands. St. Vincent Island is uninhabited, whereas Dog Island has approximately 90 private residences (mostly seasonal). The FWC prohibited possession of *Macrochelys* without a permit in 2009, and a possession permit has not been issued to any of the few residents of Dog Island. *Macrochelys* stranded on island beaches would be unlikely to travel inland over land in search of fresh water, but lakes on St. Vincent Island are connected to salt water. Oyster Pond has an outlet to the gulf on the south side of the island, whereas the other five lakes are connected to each other by small, winding channels and have an outlet near West Pass. Lake 5 (*Macrochelys* record) is the farthest from the outlet and is surrounded by dense marsh vegetation. St. Vincent Island formed during the past 4,000–5,000 years, and Dog Island during the past 3,000 years (Tanner 1988). Wetlands on St. Vincent Island have been colonized by a diverse freshwater turtle community: Florida softshell (*Apalone ferox*), snapping turtle (*Chelydra serpentina*), chicken turtle (*Deirochelys reticularia*), mud turtle (*Kinosternon* sp.), Florida cooter (*Pseudemys floridana*), and Florida red-bellied cooter (*P. nelsoni*; U.S. Fish and Wildlife Service 2012). In contrast, the only other freshwater aquatic turtle species observed on Dog Island is the mud turtle (TC, pers. obs.). Conditions for turtle colonization might have been more suitable in the past; St. Vincent Island has experienced at least two mean

sea level rises and one fall exceeding 1 m (Tanner 1988). Two adult Central American snapping turtles (*Chelydra rossignonii*) washed ashore on a Caribbean island following separate extreme weather events, which suggests that long-distance overseas dispersal may account for colonization of islands by chelydrid species (Brown et al. 2021).

Insular populations of *Macrochelys* exhibit behaviors that are rarely observed in riverine populations on the mainland. Alligator snapping turtles are seldom observed on land, except for nesting females, which usually nest within 20 m of water but sometimes as far as 200 m from water (Ewert et al. 2006). Consequently, we consider it unusual to document four *Macrochelys* on land on Dog Island during a 7-month period, particularly because only 12 full-time residents were present to detect turtles. The turtle found crawling on a road on Dog Island may have been leaving a drying wetland. Two turtles on Dog Island excavated body depressions on land. We could not find any reports of this behavior in *Macrochelys*. These partially buried turtles might have died if humans had not intervened and translocated them to fresh water.

We believe that most observations of *Macrochelys* in estuarine areas enclosed by barrier islands or peninsulas in Gulf and Franklin counties can be explained by larger turtles voluntarily entering to forage or smaller turtles being flushed from rivers during periods of high discharge. Physical conditions that affect the salinity of Apalachicola Bay include tidal elevations, wind and current velocities, precipitation, and the discharge of the Apalachicola River (Sun and Koch 2001). Water depths in the bay gently vary from 2 m near the river mouth to 4 m near ocean passes (Huang and Spaulding 2002). Surface salinity in the bay varies from about 5 ppt near the river mouth to about 30 ppt near East Pass between Dog and St. George islands; bottom salinity generally is higher than surface salinity because of vertical stratification (Huang and Spaulding 2002). Most oceanic flow into the bay enters through East Pass, and most outflow occurs through West Pass (Sun and Koch 2001). We suspect that the two *Macrochelys* found partially buried on Dog Island in November and December 2018 (Table 1) came from the Carrabelle River following significant rainfall events from Hurricane Michael in October 2018 that could have encouraged the turtles to move into St. George Sound because of reduced salinity levels. These turtles may have become osmotically stressed when salinity levels increased, and they crawled out on the closest dry land, where they dug into sand to conserve moisture. The dead adult turtles found on beaches on St. Vincent Island at West Pass and Indian Pass could have died there from osmotic water loss instead of being transported there by currents after dying elsewhere. Freshwater turtle species die within a few days of exposure to sea water (35 ppt) because of rapid body loss (Dunson 1981). Striped mud turtles (*Kinosternon baurii*) in the Lower Keys inhabit brackish wetlands at salinities <15 ppt (Dunson 1981), which is 43% sea water. This salinity level might be near the tolerance limit for freshwater turtle species (Dunson and Seidel 1986). The Apalachicola River experiences extensive seasonal variability in daily discharge ranging approximately from 250 m^3/sec in July–November to 3,000 m^3/sec in January–April (Joshi 2019). The other three observations of *Macrochelys* on estuarine beaches occurred during the latter period when peak amounts of river water are typically discharged into estuaries, reducing their salinity. Observations of two small (≤80 mm SCL) *Macrochelys* on beaches on Dog Island and Indian Peninsula and an adult in Mexico Beach in high-salinity situations are particularly surprising.

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REPEATED ATTEMPTS BY A FEMALE BOAT-TAILED GRACKLE TO FEED A ROAD-KILLED FLEDGLING

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At 1000 hours on 6 May 2018, we were traveling northeast across the causeway from Sanibel Island, Lee County, Florida, to the mainland, traversing the easternmost of the two man-made islands that comprise Sanibel Causeway Park. The two-lane highway was busy with traffic in both directions. As we neared the east end of the island, we saw a black, road-killed bird lying just out of our lane of traffic and when a car coming from the east passed, the air movement caused by the vehicle raised the bird's tail into the air and a female Boat-tailed Grackle (*Quiscalus major*) flew down to the dead bird. By that point we were close enough to see that the female had items in its mouth. The approach of our vehicle caused the female to fly away from the road, but we stopped, turned around, and parked about 20 m from the road-killed bird to confirm its identity as a fledgling male Boat-tailed Grackle. As we looked at the dead fledgling, another car passed from the east and again, the air movement caused the tail of the fledgling to flip straight up, then lay down. At that point the female Boat-tailed Grackle again flew to the bird, landed near its tail, walked around to its head, then leaned over and tried to get the dead bird to take the food it had in its mouth (Fig. 1). A car approaching from the west caused the female to fly again. Over the next 30 minutes, several cars passed by from the east, each time causing the dead grackle's tail to flip up and fall down. If a car wasn't closely approaching the dead bird from the west, the adult female grackle returned with its food and tried to feed the fledgling—only to be disturbed in its endeavor by the next approaching vehicle. We tallied seven such feeding attempts over 30 minutes.

The food in the female's bill appeared to be dog-food pellets, which Boat-tailed Grackles in Florida are known to eat (Post et al. 1996). It was certainly some pelletized human-produced food. The female Boat-tailed Grackle was hampered by severe scaly leg on both feet, which were also entangled with monofilament nylon (Fig. 2).

The roughly 4-ha easternmost island of Sanibel Causeway Park is long, narrow, and dominated by the road. Adjacent areas are mowed and shaded by a diversity of trees and shrubs. Narrow beaches on each side provide for human recreation and as depositories for tide- and wave-deposited largesse that grackles use. As part of Causeway Islands Park, the island is busy with beach-goers, picnickers, and people fishing. Birds that nest on the island must surely cope with frequent disturbance, although Post et al. (1996) suggest that such human-dominated environments may be preferred nesting areas as a result of reduced populations of predators. The heightened human activity on the island was likely responsible for use of human-provided food and entanglement of the bird's feet in monofilament fishing line—a perfect juxtaposition of the advantages and disadvantages experienced by a species living in proximity to humans.

The female grackle's stress was clearly exacerbated by the entanglement with the fishing line—and the entanglement in the line may in turn have been facilitated by the raised scales of the swollen feet caused by scaly leg. Scaly leg is known from many birds, including Boat-tailed Grackles in Florida (Pence et al. 1999), and is caused by

Figure 1. Female Boat-tailed Grackle attempting to feed her road-killed fledgling after a passing vehicle caused the tail to flip up in the air, Sanibel Causeway Park, 6 May 2018.

Figure 2. Enlargement of the female Boat-tailed Grackle's feet to show the extent of the scaly leg and the entanglement of the feet in monofilament nylon, Sanibel Causeway Park, 6 May 2018.

knemidocoptic mites (*Knemidocoptes* sp.). This female Boat-tailed Grackle had multiple reasons to be stressed, yet her nurturing instinct was incredibly strong, which one might expect because, in this species, only the female is known to feed nestlings or fledglings (Tutor 1962, Bancroft 1987, Post et al. 1996).

Parental attempts to feed a dead chick or fledgling are rare or at least rarely reported. In their review of Boat-tailed Grackle biology, Post et al. (1996) noted no reference to parental response to dead offspring. The only reference we found to a parental attempt to feed a dead chick was for an adult Yellow-vented Bulbul (*Pycnonotus golavier*) whose nest had been visited by a predator (Wee 2009). Two chicks had been in the nest and one was missing. The second was dead and on the ground beneath the nest. Neither adult returned to the nest, but at least one of them went directly to the dead chick and returned with food in its mouth to the chick at intervals throughout the morning and attempted to feed it. A major difference is that in the case of the Boat-tailed Grackle, the tail of the fledgling moved with each passing car, perhaps suggesting to the adult that her offspring was still alive.

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NEST REUSE BY THE FLORIDA SCRUB-JAY (*Aphelocoma coerulescens*) IN OCALA NATIONAL FOREST

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Most passerine birds that are cup nesters use grasses, twigs, and other plant materials to construct nests, which degrade and break down quickly and are rarely reused by the same individuals or by conspecifics (Hansell 2000). Passerines may also avoid reusing nests because old nest sites may have increased ectoparasite loads (Clark and Mason 1985, Moller 1994, Bergin 1997, Hansell 2000) or increased predation risk from predator species that revisit recently depredated nests (Sonerud 1985, Sonerud and Fjed 1987, Styrsky 2005). Despite the increased risk of predation associated with nest reuse, it is believed that several flycatcher (Tyrannidae) species reuse cup nests to save time and energy related to renesting (Briskie and Sealy 1988, Curson et al. 1996). In addition, some nest reuse by flycatchers occurs in habitats where nest sites are limited (Redmond et al. 2007, Cancellieri and Murphy 2013). In contrast, nest reuse in North American jay species (Corvidae) is extremely rare.

The Florida Scrub-Jay (*Aphelocoma coerulescens*) is a cooperative-breeding species endemic to peninsular Florida. Florida Scrub-Jays are territorial year-round, and territory size is dependent on habitat quality and group size (Woolfenden and Fitzpatrick 1984). Nesting is asynchronous, with nests built from March through June, primarily from oak (*Quercus* spp.) twigs thickly lined with tucked palmetto fibers (*Sabal etonia*; Woolfenden and Fitzpatrick 1996). Although this species readily re-nests within the same season after nest failure, and sometimes double broods, it constructs a new nest with each successive nest attempt (Woolfenden and Fitzpatrick 1984, 1996). Use of the same nest within the same breeding season is extremely rare. Woolfenden and Fitzpatrick (1996) reported two instances of nest reuse in >1,500 monitored nests (<0.1%) at Archbold Biological Station in Highlands County but provided no details. Reed Bowman also observed two instances of nest reuse in a suburban development in Highlands County where suitable nest sites appeared to be limited (R. Bowman, Archbold Biological Station, pers. comm.).

Since 2017, Florida Fish and Wildlife Conservation Commission biologists have conducted detailed nest monitoring in Ocala National Forest (ONF), the largest remaining population of the species (Stith 1999). We report here the first observation of reuse of a Florida Scrub-Jay nest in ONF and provide potential explanations for this rare phenomenon. During 2020, we discovered the earliest nest on 6 March in the building stage and the latest nest on 29 June 2020 in the nestling stage. On 31 March

2020, we discovered a Florida Scrub-Jay nest in the building stage, approximately 1.2 m above the ground located in a myrtle oak (*Q. myrtifolia*). On 17 April 2020, we observed two eggs in the nest during a routine nest check, and two unbanded adult jays were present and defensive around the nest. On 28 April 2020, we observed one egg in the nest, suggesting a partial predation. We observed an adult female jay incubating the remaining egg during three nest checks between 4 May and 11 May 2020, and we found the nest empty on 15 May 2020. The missing egg may have been infertile, resulting in one of the adult jays removing it from the nest, or it may have been removed by a predator. On 25 May 2020, we observed two new eggs in the same nest structure. Two unbanded adult jays were present and defensive around the nest. On 29 May 2020, we observed three eggs in the nest and on 3 June 2020 the nest was empty, and we assumed it had been depredated. Although this pair of Florida Scrub-Jays was unbanded, strong territoriality among Florida Scrub-Jay family groups leads us to believe that these nesting attempts likely were made by the same pair. This was the first observation of nest reuse by Florida Scrub-Jays in ONF in approximately 400 monitored nests over 5 years (K. Miller, unpublished data).

Nest reuse within or between seasons is an extremely rare behavior for North American jay species, known to occur in only two (*Cyanocitta* and *Aphelocoma*) of the five genera. These include one observation of a 1-year-old Blue Jay (*Cyanocitta cristata*) using a nest that her parents had built but abandoned 1 month prior (Laskey 1958), observations of Blue Jays reusing nests for multiple breeding seasons (Duyck and McNair 1990), and a few instances of female Mexican Jays (*Aphelocoma wollweberi*) with physical handicaps (e.g., loss of a leg) reusing a nest (McCormack and Brown 2020). We observed no evidence of either adult Florida Scrub-Jay in our study having a physical handicap.

The observation of nest reuse in ONF occurred on the eastern edge of Hughes Island, a newly designated Scrub Jay Management Area (U.S. Forest Service 2009, Hinchee and Garcia 2017) cleared and burned during 2013–2014. Scrub stands 6–7 years post-disturbance represent optimal habitat conditions for ONF and may attract Florida Scrub-Jays from older, overgrown scrub (Miller and Shea 2021). In July 2020, our post-reproductive survey (Audubon Florida 2015, Miller et al. 2015) indicated that density of family groups was nearly 20 family groups/41 ha (100 acres) within a three-territory radius of the reused nest, in contrast to the 3–7 family groups/41 ha typical elsewhere in ONF (K. Miller, unpublished data) and elsewhere in the species' range (Woolfenden and Fitzpatrick 1996). The average territory size for a Florida Scrub-Jay family group is 9 ha on the Lake Wales Ridge (Woolfenden and Fitzpatrick 1996), compared to approximately 2 ha in eastern Hughes Island in 2020. This occurrence of nest reuse may have been a function of limited quantity and quality of nesting sites within an unusually small territory in a densely populated scrub stand. Nest reuse in some kingbird species occurs more frequently when quality nest sites are rare (Redmond et al. 2007, Cancellieri and Murphy 2013). Additional hypotheses for nest reuse include time and energy savings through not needing to construct a new nest combined with seasonal pressures from late-season nesting or inexperienced jays. Both hypotheses were noted as insignificant in other passerine species known to reuse nests within years (Redmond et al. 2007, Cancellieri and Murphy 2013), but we cannot rule out those hypotheses with the limited information we have on the nesting jays.

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ATTEMPTED PREDATION OF A BLACK SWAMPSNAKE (*Liodytes pygaea*) BY A GLOSSY IBIS (*Plegadis falcinellus*) IN ORANGE COUNTY, FLORIDA

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Glossy Ibis (*Plegadis falcinellus*) are generalist predators with seasonally variable diets that primarily consume aquatic invertebrates and plant matter, occasionally taking vertebrates when available (Kushlan 1979, Acosta et al. 1996, Avila et al. 2009). Insects and their larvae (beetles, heteropterans, dragonflies and damselflies, grasshoppers and crickets, dipterans, caterpillars, and others), molluscs, crustaceans, annelids (worms and leeches), and arachnids are commonly recorded invertebrate prey, while vertebrates most commonly taken include fishes, frogs, and salamanders (Le Souëf 1917, Marchant and Higgins 1990, Hancock et al. 1992, Petrescu 1999, Macías et al. 2004). Though often included in reviews of Glossy Ibis diet, published records of snake predation by this species are rare. Baynard (1912:168, 1913:109) reported young "moccasins" as prey of the Glossy Ibis in Florida; presumably these records refer to the Florida cottonmouth (*Agkistrodon conanti*) but might also represent misidentified water snakes (*Nerodia* spp.) or other species. Baynard (1913) noted that snakes were a major component of the diet of nesting and young Glossy Ibis they studied; four chicks reportedly consumed more than 100 snakes along with several thousand invertebrates between ages 19–30 days. Unfortunately, except for the aforementioned "moccasins," Baynard (1912, 1913) did not identify any prey items to species level. To our knowledge, these records are the only published evidence of snakes as natural prey items of Glossy Ibis. Here, we document the attempted predation of a black swampsnake (*Liodytes pygaea*) by a Glossy Ibis.

On 6 February 2021 at approximately 1100 EST, SDW observed and photographed an adult Glossy Ibis attacking an adult black swampsnake along the Lake Apopka Wildlife Drive in Orange County, Florida, USA (28.669°N, 81.576°W; WGS84). The ibis held the posterior third of the snake's body firmly in its bill while the snake twisted and flailed, occasionally wrapping itself around nearby vegetation (Fig. 1). At least once, the ibis released and then re-secured the snake on the anterior portion of the body closer to the head, while the snake continued its attempts to escape. Eventually, the observation was interrupted by additional vehicular traffic, so whether the ibis ultimately consumed the snake is not known. The events witnessed in this encounter are similar to a published observation of a black swampsnake preyed upon by a White Ibis (*Eudocimus albus*; Durso et al. 2017).

Black swampsnakes are secretive, aquatic natricines that range from peninsular Florida to coastal North Carolina in the southeastern United States (Ernst and Ernst 2003). Until recently, records of avian predation on this species were limited; documented predators now include Great Egrets (*Ardea alba*), American Bitterns (*Botaurus lentiginosus*), Red-shouldered Hawks (*Buteo lineatus*), Green Herons (*Butorides virescens*), and White Ibis (Durso and Rosenthal 2016, Durso et al. 2017). McEwan and Hirth (1980) discovered black swampsnake remains in a Bald Eagle (*Haliaeetus leucocephalus*) nest, though Durso and Rosenthal (2016) suggested those remains might represent a secondary record from the stomach of a predatory fish consumed by the

Figure 1. A Glossy Ibis (*Plegadis falcinellus*) attempting to prey upon a black swampsnake (*Liodytes pygaea*) in Orange County, Florida. Photograph by Steven D. Wilkinson.

eagles. Finally, Dowling (1950:31) noted an observation of a Great Blue Heron (*Ardea herodias*) consuming a "small, smooth-scaled snake, either *Seminatrix* or *Liodytes*," though whether the heron was consuming a black swampsnake (*L.*[=*Seminatrix*] *pygaea*) or a striped crayfish snake (*L. alleni*) was not clear. Black swampsnakes in Florida are nocturnally active year-round but shelter under aquatic vegetation during the day or in colder temperatures (Dowling 1950, Ernst and Ernst 2003). Glossy Ibis, meanwhile, usually forage in shallow muddy marshes with short vegetation (Hancock et al. 1992). Although we did not observe the initial moments of this interaction, it is plausible the ibis encountered the sheltering snake while actively probing vegetation and debris with its bill.

Snakes represent notably larger prey items than the invertebrates typically consumed by Glossy Ibis. Frederick and Bildstein (1992) reported 99% of prey items taken by Glossy Ibis in Venezuela—primarily arthropods, annelid worms, and small gastropods—were smaller than 20 mm. In contrast, Winne et al. (2005) reported snout-vent lengths (SVL) between 218–356 mm in adult male black swampsnakes, 212–446 mm in adult females, and 125–211 mm in immature juveniles, with body masses ranging from 4.8–38.0 g for adults and 1.6–7.0 g for immature snakes. Because we observed the snake in this interaction from a distance and we did not handle or sex the snake, we can only estimate the snake's length; it appeared to be an adult at least twice as long as the ibis's bill (average bill length: males = 131 mm; females = 104 mm; Hancock et al. 1992), suggesting a conservative estimate of total length between 200–300 mm. This size is notable, but not outside of the limits of other vertebrates taken by Glossy Ibis. For example, sharp-ribbed newts (*Pleurodeles waltl*) are known prey of Glossy Ibis in Spain (Macías et al. 2004) and may reach maximum lengths of 300 mm on the Iberian Peninsula (Beukema et al. 2013); young cottonmouths—presumably the small "moccasins" recorded by Baynard (1912, 1913)—may be upwards of 280 mm SVL (Blem and Blem 1995, Ernst

and Ernst 2003). Still, a prey item of this size is noteworthy because Glossy Ibis rarely take prey larger than a few centimeters (Frederick and Bildstein 1992, Petrescu 1999).

The paucity of literature on snakes as prey items of Glossy Ibis raises the question of whether these birds avoid feeding on snakes except under specific circumstances, or if they frequently attempt to prey upon snakes but with low success; either could account for the lack of snakes recorded in diet studies, especially those based on gut, fecal, or regurgitate contents. Kushlan (1979) showed that White Ibis will selectively release larger potential prey as they are associated with higher robbing rate from other birds, though they are known to consume snakes (Dorn et al. 2011, Durso et al. 2017). Like White Ibis, Glossy Ibis are also tactile feeders (Hancock et al. 1992) and thus might generally release larger potential prey except in specific circumstances. Taking into account the fact that in the many years between Baynard's (1912, 1913) reports and our observation there appear to be no published records of Glossy Ibis preying upon snakes, it seems most likely that these birds avoid feeding on snakes except under very specific circumstances. Notably, both our and Baynard's (1912, 1913) observations occurred in Florida, and snake predation has not been reported elsewhere in the range of the Glossy Ibis; though this trophic interaction likely occurs more commonly than is documented, the circumstances that give rise to it are in need of further study.

We provide the first record of a Glossy Ibis attempting to prey upon a black swampsnake. As reports of snake predation by Glossy Ibis are rare, our record provides valuable new data on this interaction and adds a new species to the list of potential avian predators of black swampsnakes. As noted by Durso et al. (2017), the frequency of aquatic snake predation by wading birds together with the resulting effects on population, community, and ecological dynamics should be investigated further.

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ECTOSYMBIONTS FROM A NORTHERN MOCKINGBIRD ON VACA KEY, FLORIDA

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The Northern Mockingbird (*Mimus polyglottos*) is a year-round resident in Florida and is declining in numbers (Hanauer et al. 2010, Stracey and Robinson 2012). One specimen was found dead in the City of Marathon (Vaca Key) on 13 January 2021, similar to another specimen previously encountered (Hribar 2019). It is unknown how long the bird had been dead, but it died sometime between the previous evening and earlier that morning. I washed the bird, collected ectosymbionts, and mounted ectosymbionts on microscope slides using the same methods in prior reports (Hribar and Miller 2011).

I recovered approximately 50 feather mites. I mounted 34 mites on slides and preserved the remainder in 95% ethanol. Dr. Sergei Mironov of the Zoological Institute of the Russian Academy of Sciences, St. Petersburg examined the mites. He identified four species, two in the family Proctophyllodidae, one in the family Analgidae, and one in the family Xolalgidae. An undescribed species of *Metapterodectes* (Proctophyllodidae; 7 males, 11 females, 3 tritonymphs) was the most numerous of the four mite species. Morphologically this species is very close to *Metapterodectes saturninus* Hernandes, 2016 from the Chalk-browed Mockingbird (*Mimus saturninus*) in Brazil, and *Metapterodectes toxostomae* Mironov and OConnor, 2014 from the Brown Thrasher (*Toxostoma rufum*; Mironov and OConnor 2014; Hernandes et al. 2017). He identified one male and two females of another undescribed proctophyllodid species in the genus *Proctophyllodes*. Finally, he identified an undescribed species of *Analges* (7 males, 3 females, 1 tritonymph; Analgidae), and an undescribed species of *Xolalgoides* (2 males, 2 females; Xolalgidae). Proctophyllodid mites in the genera *Proctophyllodes* and *Metapterodectes* inhabit the flight feathers (remiges and rectrices) of their hosts, whereas mites of the families Analgidae and Xolalgidae inhabit down feathers and downy parts of the contour feathers (Mironov and González-Acuña 2011, Mironov 2012).

I collected, cleared, and slide-mounted two biting midges (Diptera: Ceratopogonidae). Dr. William Grogan of the Florida State Collection of Arthropods, Gainesville subsequently examined the midges. He identified these two females as nr. *Forcipomyia* (*Lepidohelea*) *eadsi* Wirth and Spinelli but concluded they differ from that species in having globular spermathecae with broad necks and wing lengths of only 0.73–0.74 mm. *Forcipomyia eadsi* is known from Monroe County (Hribar and Grogan 2005, Grogan et al. 2010) and differs from the two collected females from the mockingbird in having elongate ovoid spermathecae with slender necks and wing lengths 0.82–1.00 mm.

I also collected a springtail (Collembola: Entomobryidae) from the mockingbird. Dr. Felipe Soto-Adames of the Florida State Collection of Arthropods, Gainesville, examined the specimen and identified it as *Seira dowlingi* (Wray). *Seira dowlingi* is one of the most widely distributed collembolan species in the Americas. It is known from Brazil, Peru, Dominican Republic, Puerto Rico, Cuba, Ecuador, Panama, and the United States (Arizona and Illinois; Cipola et al. 2019). The genus *Seira* includes epiedaphic (i.e., living on the soil surface) springtails found mostly in litter; they can also be found in vegetation and in the canopy and even on uncovered top soils (Christiansen and Bellinger 2000). This specimen is the first of its species reported from Monroe County, Florida.

It is one thing to report collections of ectosymbionts; it is another matter to attempt to explain their presence in time, place, and upon a given host. The recovery of undescribed feather mite species is not surprising. There are probably thousands of species in the world fauna of feather mites that await detection and description (Dabert 2004). An undescribed *Analges* species was previously recovered from a Northern Mockingbird on Vaca Key; the mites reported herein appear to be only the second records of feather mites from a Northern Mockingbird in Florida (Hribar 2019). At present it is not known whether the two mite samples are conspecifics. The finding of two *Forcipomyia* midges is more interesting. Although females of *Forcipomyia* in the subgenus *Lasiohelea* feed on vertebrate blood (Borkent and Spinelli 2007), females in the subgenus *Lepidohelea* have reduced, non-functional mandibles. Behavior of adult *F. eadsi* is unknown, so there is no good explanation for why the midges were found in the feathers (Wirth and Spinelli 1993). It is possible that the springtail was transported from elsewhere by the mockingbird. Springtails in the families Entomobryidae and Poduridae are known to inhabit birds' nests (Hicks 1953) so it is possible that the mockingbird acquired the springtail from a nest. However, the breeding season for the Northern Mockingbird in Florida starts in late February, so this is unlikely (Derrickson and Breitwisch 1992). Furthermore, Hicks (1953) did not report springtails from mockingbird nests. The Northern Mockingbird is non-migratory except in the extreme northern parts of its range, so it is also unlikely that the bird brought the springtail from elsewhere (David et al. 1990). It may be that the springtail species was simply previously undetected in Monroe County and that a specimen crawled onto the dead bird.

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TREASURER'S REPORT - 2020

Fund	2020 Income and Expenses	Amount
Operating Fund		
Income		
	Dues	$8,410.00
	Unrestricted Gifts	$2,068.44
	Interest/Asset Appreciation	$11,041.96
	Shirt/Hat Sales	$109.00
	Page Charges	$875.00
	Meeting Income	$0
	Total Income	**$22,504.40**
Expenses		
	FFN Printing	$11,070.15
	Postage & Shipping	$0
	Operating Expenses	$1,436.38
	Website Costs	$0
	Shirt Sales Costs	$0
	FL BBA2	$0
	Meeting Expenses	$0
	Records Committee	$0
	Festival Sponsor-FL Birding & Nature Fest	$0
	Florida Keys Hawkwatch-Sponsor	$350.00
	Total Expenses	**$12,856.53**
Surplus		$9,647.87
Special Publications Fund		
Income		
	Sales	$193.00
	Interest/Asset Appreciation	$8,876.48
	Total Income	**$9,069.48**
Expenses		
	Postage & Shipping	$46.73
	Total Expenses	**$46.73**
Surplus		$9,022.75
Cruickshank Awards Fund		
Income		
	Interest/Asset Appreciation	$8,951.89
	Gifts	$35.00
	Total Income	**$8,986.89**
Expenses		
	Research Award	$4,250.00
	Education Grant	$0
	Total Expense	**$4,250.00**
Surplus		$4,736.89

(Continued)

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Fund	2020 Income and Expenses	Amount
Robertson Awards Fund		
Income		
	Interest/Asset Appreciation	$4,257.94
	Gifts	$210.00
	Total Income	**$4,467.94**
Expenses		
	Robertson Award	$0
	Total Expense	**$0**
Surplus		$4,467.94
Endowment Fund		
Income		
	Interest/Asset Appreciation	$6,795.46
	Gifts	$10.00
	Total Income	**$6,805.46**
Surplus		$6,805.46
Friends of FFN Fund		
Income		
	Interest/Asset Appreciation	$1,153.65
	Gifts	$1,200.00
	Total Income	**$2,353.65**
Expenses		
	Color Printing	$1,000.00
	Total Expense	**$1,000.00**
Surplus		$1,353.65
Student Travel and Awards Fund		
Income		
	Interest/Asset Appreciation	$1,035.41
	Gifts	$60.00
	Total Income	**$1,095.41**
Expenses		
	Student Awards and Travel Cost	$0
	Total Expense	**$0**
Surplus		$1,095.41
Net Income		$37,129.97

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